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Diesel Particulate Study

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16. ABSTRACT

The purpose of th~~is~~e study is to provide a comprehensive assessment of the costs and environmental effects of the control of diesel particulate emissions and to recommend a regulatory strategy for their control. As such, this study expands, updates and combines the Regulatory Analyses supporting the light-duty diesel particulate final rule and the heavy-duty diesel particulate (proposed rule. The study identifies current and future diesel particulate) emissions and exposure levels, assess the health and welfare impact of diesel particulate, and estimate the costs of controlling diesel particulate emissions to various levels. The study also integrates these aspects of diesel particulate control and will develop, evaluate, and recommend a regulatory control strategy.

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Part I

INTRODUCTION

I. Purpose of the Study

EPA's study of the costs and environmental effects of the control of diesel particulate emissions and their regulation has been underway for some time. Emission standards for 1982 and later diesel-powered light-duty vehicles and light-duty trucks (light-duty diesels) were promulgated in 1980. Similar standards for diesel-powered heavy-duty diesels were proposed in 1981.

The pertinent data on both the costs and benefits of diesel particulate control have been constantly changing over time. This is particularly true of the last two to three years since the time of the rulemakings mentioned above. Emission control technology has been constantly evolving, changing both baseline emission rates and the ability and costs of further control. In addition, a number of cancer-related health studies on diesel particulate have been completed in the last two years, allowing an assessment of benefits in this area which was not previously possible. Data and projections in other key areas have also been changing, resulting in a need for EPA to reexamine its regulatory position.

Recent regulatory activity by EPA has reflected these changing circumstances. In the light-duty area, EPA has promulgated a delay of the more stringent, 1985 standards until 1987, leaving in place the current 1982 standards through 1986. This action is based on the fact that a new control technology could not be applied fleet-wide for the 1985 model year, but will require two additional years of effort. This new technology is referred to as a trap-oxidizer and produces substantial reductions (i.e., greater than 50 percent) in diesel particulate emissions. In the heavy-duty area, EPA has announced its intention to repropose its particulate standard along with the NOx standard proposal, because of the interelationship between NOx and particulate control. This combined proposal should enable this interaction between the two pollutants to be better assessed and facilitate a more orderly standard setting process.

The purpose of this study is to provide a comprehensive assessment of the costs and environmental effects of the control of diesel particulate emissions and to recommend a regulatory strategy for their control. As such, this study expands, updates and combines the Regulatory Analyses supporting the light-duty diesel (LDD) particulate final rule and the heavy-duty diesel (HDD) particulate proposed rule.[1,2] The study will identify current and future diesel particulate emissions and exposure levels, assess the health and welfare impact of diesel particulate, and estimate the costs of controlling diesel particulate emissions to various levels. The study will then integrate these aspects of diesel particulate control and develop, evaluate, and recommend a regulatory control strategy.

This study does not attempt to economically quantify the changes in health or welfare that are associated with various diesel particulate control strategies. Such an economic quantification of benefits is beyond the scope of this report, but has been performed for various diesel particulate control strategies under contract to EPA.[3] The reader should consult this document and others in the literature for information on the economic benefits of controlling diesel particulate.

Four regulatory scenarios are examined in this report which cover a wide range of technological stringency. The least stringent control scenario (the relaxed scenario) would require no further control from LDDs and only very modest reductions from HDDs, representing the least stringent degree of control conceivable. The next most stringent scenario (the intermediate scenario) would require the application of advanced non-trap technology. By their nature, these techniques are quite cost effective and this scenario represents a modest degree of control that should be available at low cost. The third scenario (the base scenario) consists of the current trap-based standards (i.e., the 1985 LDD particulate standards and that proposed for 1986 HDDs). This scenario provides more control than that achievable through non-trap technology, but will be more costly and less cost effective than the second scenario due to the use of trap technology. The fourth scenario (the stringent scenario) represents the greatest degree of control presently conceivable. Nearly all vehicles would be equipped with traps under this scenario. These scenarios are summarized in Table 1.

As an averaging concept has already been promulgated for compliance with the 1987 LDDV and LDDT trap-based particulate standards, the flexibility it provides will be presumed here for the base, intermediate, and stringent scenarios. As it is likely, but not certain, that a similar program will be proposed for HDDs, both averaging and non-averaging situations will be examined for the base scenario. Only averaging situations will be considered for the HDD intermediate and stringent scenarios. However, under the HDD stringent scenario, all vehicles are likely to require traps, so averaging does not add much flexibility.

Table 1

Emission Control Scenarios*

	Particulate Standards		
	Level*	Implementation Date**	Range of NOx Standards
Relaxed Scenario:			
LDDV (g/mi)	Current Levels(NA)	--	1.0, 1.5, 2.0
LDDT (g/mi)	Current Levels(NA)	--	1.2, 1.7, 2.3
HDDV (g/BHP-hr)	0.6(NA)	1988	--
Intermediate Scenario:			
LDDV (g/mi)	0.30(A)	1987	1.0, 1.5, 2.0
LDDT (g/mi)	0.35(A)	1987	1.2, 1.7, 2.3
HDDV (g/BHP-hr)	0.40(A)	1988	--
Base Scenario:			
LDDV (g/mi)	0.20(A)	1987	1.0, 1.5, 2.0
LDDT (g/mi)	0.26(A)	1987	1.2, 1.7, 2.3
HDDV (g/BHP-hr)	0.25(NA & A)	1988	--
Stringent Scenario:			
LDDV (g/mi)	0.08(A)	1987	1.0, 1.5, 2.0
LDDT (g/mi)	0.105(A)	1987	1.2, 1.7, 2.3
HDDV (g/BHP-hr)	0.10(N/A)	1988	--

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* (A) means averaging program available, (NA) means no averaging program available. Also, all control scenarios were evaluated for their incremental effects beyond continuing the relaxed scenario.

** The implementation dates for the various HDD standards analyzed in Part II were arbitrarily chosen at 1988 for simplicity. In reality, as discussed in Part I most of these standards would likely be implemented about 1990, except for the 0.6 g/BHP-hr standard which appears feasible by 1987.

A factor that must be considered when assessing the ability to control diesel particulate is the level of the applicable NOx standard. In general, as NOx emissions are reduced, engine-out levels of particulate increase. Thus, under a stringent NOx standard the technically feasible level of particulate control (without the use of aftertreatment devices) will be higher than under a lenient NOx standard. Because there is currently some doubt as to what the NOx standards will be in the years covered by this study, three different LDDV standards were evaluated: 1) 1.0, 2) 1.5, and 3) 2.0 g/mi. As the NOx standard for LDDTs is directly influenced by that for LDDVs, this study will also evaluate the effect of the three LDDT NOx standards equivalent to those for LDDVs: 1.2, 1.7, and 2.3 g/mi.

The question of the appropriate NOx standard for HDDs is dealt with in a more straightforward fashion than LDDs, since the level of the standard will be set by EPA. While Section 202(a)(3)(A)(ii) of the CAA requires a NOx standard of 1.7 g/BHP-hr, this level is not feasible for HDDs. Thus, EPA must set a revised NOx standard under the requirements of Section 202(a)(3)(B), which are very similar to the requirements specified for the HDD particulate standard in Section 202(a)(3)(A)(iii). Thus, under all scenarios, the HDDV NOx standard is treated as a variable and identified in much the same way as the particulate standard.

II. Organization of the Study

The study has been segregated into two parts. Part I of the study contains this introduction and an overall evaluation of control options. Part II contains the supporting technical analyses.

In addition to describing the context, purpose and organization of the study, this introduction describes the control scenarios evaluated and the diesel sales projections used throughout the analysis. The following "Evaluation of Control Options" summarizes the costs and environmental benefits of the various diesel particulate control scenarios and then goes on to compare and evaluate their relative strengths and weaknesses.

The supporting technical analysis is contained in ten chapters. The first seven chapters address the benefits of control, including vehicular emissions (Chapter 1), nationwide and urban emissions (Chapter 2), air quality and exposure (Chapter 3), visibility (Chapter 4), carcinogenic risk (Chapter 5), non-carcinogenic health risk (Chapter 6) and soiling (Chapter 7). Two chapters address the cost (Chapter 8) and cost effectiveness (Chapter 9) of control, respectively. The last chapter (Chapter 10) addresses the sensitivity of the technical results to key assumptions made throughout the study and includes other, more secondary types of analyses.

The primary technical analyses (i.e., those contained in Chapters 1 through 9) will evaluate only the relaxed and base control scenarios because these two scenarios are considered the most likely to occur and all technical concepts associated with the stringent scenario, such as control technology, are also contained in the base scenario. The NOx standards of the main analysis will be 1.5 and 2.3 g/mi for LDDVs and LDDTs, respectively, because they are the current standards and certification test data is available under these levels. As the methodology used in Chapter 1 to adjust particulate emissions for different NOx emissions standards is subject to some error, this will minimize the use of such adjustments in the main analysis. The primary technical analyses (i.e., Chapters 2 through 10) also evaluate two diesel sales projections: the best and worst estimates. The impact of: 1) other LDDV and LDDT NOx standards on the relaxed and base particulate control scenarios, 2) the stringent control scenario under all NOx standards, and 3) the effect of a no growth diesel sales projection will be evaluated in Chapter 10, Sensitivity. As little firm data is available concerning the technology associated with the intermediate sceanario, it will also be addressed in Chapter 10.

EVALUATION OF CONTROL OPTIONS

I. Introduction

The purpose of this evaluation is to first combine the results of supporting technical analyses, which addressed in detail the costs and environmental effects of controlling diesel particulate emissions, and then conduct a comprehensive comparison of the feasible control scenarios in light of their effect. Three stages are necessary to do this.

First, the health and welfare effects of the various particulate control scenarios will be assessed to identify the most significant impacts and to determine the need for control. In addition, this first step will also identify the key aspects of the cost and cost effectiveness of diesel particulate control to form a basis for assessing the degree of control which is reasonable. This first step of the analysis will focus on the most likely set of external factors for light-duty diesels (LDDs), such as the best estimate diesel sales projections and the oxides of nitrogen (NOx) standards currently mandated by the Clean Air Act (CAA) (1.0 gram per mile (g/mi) for light-duty diesel vehicles (LDDVs) and an equivalent standard (1.2 g/mi) for light-duty diesel trucks (LDDTs)). The impact of various conceivable sales projections for heavy-duty diesels (HDDs) is not significant and will not be considered in this chapter.

Second, viable strategies for controlling diesel particulate emissions will be identified. While the base, relaxed, and stringent control scenarios are addressed throughout the supporting technical analyses as likely control scenarios (see Table 1 for a description of these scenarios), the number of such scenarios was kept to a minimum to retain some measure of control over the scope of the technical analysis. Here, the levels of control contained in the three primary scenarios for the various vehicle groups will be combined in various ways and viable intermediate levels of control identified. The relative feasibility of these alternative control strategies along with their costs, if not already addressed in the supporting technical analyses, will then be discussed.

Third, the significant features of the various control strategies identified in the second stage will be compared and evaluated. Those strategies whose disadvantages far outweigh their advantages will be discarded, leaving only the most viable strategies for final consideration. The effect of the various external factors on LDD control will also be evaluated at this stage.

Table 1

Description of Control Scenarios Analyzed in the Supporting Technical Analyses[1]

	Relaxed Scenario	Base Scenario	Stringent Scenario
LDDV[2]	Current Technology	0.2 g/mi	0.08 g/mi
LDDT[3]	Current Technology	0.26 g/mi	0.105 g/mi
HDDV	0.6 g/BHP-hr	0.25 g/BHP-hr	0.1 g/BHP-hr

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[1] The availability of emissions averaging was assumed throughout for LDDVs and LDDTs, while both averaging and non-averaging situations were examined for HDDVs in the supporting technical analyses.

[2] The three control scenarios for LDDVs are each evaluated under three NOx control scenarios in the supporting technical analyses: 1.0 g/mi, 1.5 g/mi, and 2.0 g/mi.

[3] The three control scenarios for LDDTs are each evaluated under three NOx control scenarios in the supporting technical analyses: 1.2 g/mi, 1.7 g/mi, and 2.3 g/mi.

II. Key Costs and Environmental Effects of Controlling Diesel Particulate

A. Benefits

Chapters 2 through 7 and 10 of the supporting technical analyses address the environmental impacts associated with diesel particulate emissions. Four distinct health or welfare effects were addressed: 1) non-cancer health effects, 2) carcinogenic health effects, 3) visibility, and 4) soiling. These four effects are addressed below.

1. Non-Cancer Health Effects

The assessment in Chapter 6 was not able to make a firm finding with respect to the non-cancer health effects of human exposure to suspended diesel particulate. Nevertheless, two general conclusions were made. First, it is possible that the organic compounds adsorbed onto diesel particulate may be more hazaradous than general suspended particulate matter less than 10 micrometers in diameter (PM_{10}), but the available scientific evidence was inconclusive on this issue. Second, there was some very limited information suggesting that diesel particulate may be less hazardous under certain conditions (i.e., month breathing) than certain subfractions of PM_{10}. Here again, however, no conclusive judgment could be made on this point. The overall decision was to treat diesel particulate like any other type of particulate less than 10 micrometers in diameter with respect to non-cancer health effects. This decision leads to an ability to use the air quality and exposure estimates in conjunction with projected compliance with the National Ambient Air Quality Standards (NAAQS) for PM_{10} as an indicator of the impact of diesel particulate emissions on non-cancer public health.

Table 2 contains the air quality and exposure estimates for the various diesel control scenarios under best estimate sales (and the 1.0/1.2 g/mi NOx standards for LDDVs and LDDTs). The estimates for these scenarios were developed by applying the ratio of the urban emission estimates of Chapters 2 and 10 to the air quality and exposure estimates of Chapter 3. Three indicators of impact are shown: 1) ambient urban concentrations, 2) microscale concentrations, and 3) annual average urban exposures. It can reasonably be argued that the microscale impacts are the least critical. They represent very short-term exposures (a few minutes to one hour) and exposures of this length and at these levels would not be expected to cause acute health effects. The remaining two indicators both have relative advantages and disadvantages. The ambient urban concentrations include a wide range of city sizes and

Table 2

Indicators of Non-Cancer Health Effect Impacts

		1995		
	1980	Relaxed	Base	Stringent
Lead-Based Ambient Diesel Particulate Concentrations (ug/m^3)*				
City Population				
Greater than 1,000,000	1.2-2.7	3.1-7.4	1.6-3.9	0.9-2.3
500,000-1,000,000	0.8-1.8	2.2-4.9	1.2-2.5	0.7-1.5
250,000-500,000	0.9-1.5	2.4-4.1	1.3-2.1	0.8-1.2
100,000-250,000	0.6-1.6	1.6-4.3	0.7-2.2	0.4-1.3
Microscale Diesel Particulate Concentrations (ug/m^3)				
Roadway Tunnel				
Typical	57	122	63	36
Severe	145	309	160	93
Street Canyon				
Typical	2	5	2	1
Severe	14	30	16	9
On Expressway				
Typical	6	13	7	4
Severe	26	55	28	16
Beside Expressway	5	11	6	4
Annual Average Exposure to U.S. Urban Dwellers (ug/m^3)				
Total	2.2	6.0	3.1	1.8

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* Ranges are average values plus and minus one standard deviation.

meteorological conditions, but only represent the ambient concentration at one or two particular monitor sites in each city. The annual average urban exposures include a wide range of individual activity pattern effects, but overall are only based on exposures in four U.S. cities. Because the two modeling approaches were shown to be generally consistent in Chapter 3 and the exposure estimates are more simple to describe, they will receive the primary emphasis here.

As can be seen from Table 2, then, the estimate of annual average urban exposure ranges from 1.8 to 6.0 ug/m^3 depending on the scenario chosen. These are significant levels compared to the levels of the revised NAAQS for PM$_{10}$ that the Agency is currently considering (approximately 55 ug/m^3). Even after post-1983 expenditures of nearly $2 billion (1982 Net Present Value), roughly 100-150 areas would still not be in attainment of such an NAAQS. A rough estimate would be that an additional 4 ug/m^3 in ambient particulate burden would cost an additional $44-81 million per year (1982 dollars) in other than mobile source attainment costs and result in an additional 13-33 non-attainment areas. Thus, the exposure impacts shown in Table 2 are measurable and argue for further control of diesel particulate emissions.

It is also useful to examine the source of these impacts. Table 3 shows the distribution of urban emissions in 1995 between four categories of diesels: LDDVs, LDDTs, medium-duty vehicles/light heavy-duty vehicles (MDV/LHDVs) (i.e., Classes IIB-VI) and heavy heavy-duty vehicles (HHDVs) (i.e., Classes VII-VIII). As can be seen, the two light-duty categories contribute roughly half the emissions under all the scenarios, as do the two heavy-duty categories. Thus, control is needed from both general categories if the ambient levels and exposures described in Table 2 (and the other impacts described below) are to be reduced to the furthest degree feasible.

2. Carcinogenic Health Effects

The results of the carcinogenic health effects analysis of Chapter 5 are shown in Table 4 for the three diesel control scenarios. As was done in the previous section, the urban emission estimates of Chapters 2 and 10 were used to determine the cancer risk estimates for the stringent scenario and the other scenarios with stringent light-duty NOx standards. Due to the basic uncertainties inherent in attempting to predict cancer risk, values in Table 4 should be regarded only as best estimates of risk relative to other proven carcinogens. Because of limited studies on the human carcinogenic potential of diesel particulate, estimates are based primarily on results of clinical tests performed on animals and lower organisms.

Table 3

1995 Urban Diesel Particulate Emissions (metric tons)

	Scenario		
	Relaxed	Base	Stringent
LDDV	36,800 (27%)*	19,700 (27%)	10,700 (26%)
LDDT	21,800 (16%)	11,700 (16%)	5,900 (14%)
MDV/LHDV	9,400 (7%)	4,700 (7%)	2,700 (6%)
HHDV	67,600 (50%)	36,100 (50%)	22,600 (54%)
TOTAL	135,600	72,200	41,900

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* Figures in parentheses indicate percent of total.

Table 4

Comparison of Risks from Various Sources

Sources of Risk	Estimated Annual Risk (risk/person-year)	Exposed Population
Commonplace Risks of Death		
Motor Vehicle Accident	222.0×10^{-6}	Entire U.S.
Drowning	26.0×10^{-6}	Entire U.S.
Burns	21.0×10^{-6}	Entire U.S.
Tornados, Floods, Lightning, Tropical Cyclones and Hurricanes	2.0×10^{-6}	Entire U.S.
Risks of Cancer Incidence		
Diesel Particulate (1995):		Urban U.S.
Relaxed Scenario	$1.6 \times 10^{-6} - 8.4 \times 10^{-6}$	
Base Scenario	$0.8 \times 10^{-6} - 5.1 \times 10^{-6}$	
Stringent Scenario	$0.5 \times 10^{-6} - 3.0 \times 10^{-6}$	
Natural Background Radiation (sea level)	20.0×10^{-6}	Entire U.S.
Average Diagnostic Medical X-Rays in the United States	20.0×10^{-6}	Widespread
Frequent Airline Passenger (4 hours per week flying)	10.0×10^{-6}	Limited
Four Tablespoons Peanut Butter Per Day (due to presence of aflatoxin)	8.0×10^{-6}	Fairly Widespread
Ethylene Dibromide	4.2×10^{-6}	Widespread
One 12-Ounce Diet Drink Per Day	2.6×10^{-6}	Widespread
Arsenic	1.7×10^{-6}	1% of U.S.
Miami or New Orleans Drinking Water (due to presence of chloroform)	1.0×10^{-6}	Southern U.S., Urban
Lung Cancers:		Entire U.S.
For Smokers Due to Smoking	419.0×10^{-6}	
For General Population Due to Causes Other Than Smoking	73.9×10^{-6}	

As can be seen, the risk of contracting lung cancer from exposure to diesel particulate appears small compared to many of the other risks shown in Table 4. Yet the upper limits of the diesel risk estimates under all control scenarios would still represent 4-11 percent of all non-smoking related lung cancer in the U.S. Even the lower limit of risk under the relaxed scenario is above one out of a million, which has been used by regulatory agencies in the past as a yardstick for determining the need to regulate. Thus, while the estimated cancer risk is not tremendous, it is nonetheless measurable and suggests the need for some degree of control.

3. Visibility Impacts

The visibility impacts of the various diesel control scenarios are presented in Table 5. The figures were generated from the results of Chapters 2, 4, and 10 in the same manner as the air quality and exposure estimates of Table 2. The assumption that visibility impact is proportional to emissions does not apply as readily as it does for air quality impact and exposure, but the resultant inaccuracy is insignificant for the purposes of this report.

As can be seen, the most stringent control scenario provides a 2-16 percent improvement in visibility over the least stringent scenario. While the upper limit of this impact is probably perceptible, it is more difficult to tell about the lower limit. Differences between intermediate control strategies would be much less than these absolute effects. Thus, while the overall impact of diesel particulate on visibility could be characterized as perceptible and deserving of control, the change in visibility between the individual control strategies is difficult to characterize based only on an analysis of physical effects.

No attempt is made in this study to quantify the visibility impacts into a dollar value. This is not meant to imply that even a small degradation in visibility due to diesel particulate is not economically important. Such a benefits analysis simply is not within the scope of this report.

4. Soiling Impacts

As was evident from the soiling analysis in Chapter 7, little can be said about the physical effects of diesel particulate on soiling based on a review of the scientific literature. It is possible, however, that diesel particulate, gram- for-gram, may have a disproportionate effect on soiling when compared to other types of particulate. Otherwise, little information on soiling is available in the scientific

Table 5

Average Reduction in Visibility Due to Diesel Particulate in 1995 (percent)

City Size (population)	Relaxed	Base	Stringent
More than 1,000,000	24	15	9
500,000-1,000,000	10	6	4
250,000-500,000	7	4	2
100,000-250,000	5	2	1

literature. Thus, no conclusion can be made concerning the need to control diesel particulate based only on the physical effects of soiling. A decision on this issue may be possible based on a review of the economic literature, but such a benefits analysis is beyond the scope of this report.

5. Overall Evaluation

A number of conclusions can be drawn from the scientific analysis of the four health and welfare effects of diesel particulate emissions. One, the effect of diesel particulate emissions on visibility and both carcinogenic and non-cancer health effects are noteworthy, while the soiling effect is for the most part unknown. Two, while these impacts are measurable and argue for control, this analysis could not readily discern the level of control at which these effects disappear and where control is clearly sufficient. This is particularly true of the cancer risk impact. Thus, while this analysis shows the need for more control, it cannot alone be used to demonstrate when sufficient control has been applied. In the absence of a precise cost benefit analysis, such a decision can be made, albeit less precisely, by considering the cost and cost effectiveness of control. The groundwork for this is laid in the next two sections.

B. Costs

The economic analysis performed in Chapter 8 identified three measures of economic impact: 1) trap-oxidizer cost per vehicle (coupled with a percentage of vehicles requiring traps), 2) diesel sales reductions, and 3) nationwide annual costs (or 5-year costs). These three measures are shown in Tables 6-8 for the two more stringent control scenarios relative to the relaxed scenario.

Table 6 shows both the cost to the consumer of equipping a vehicle with a trap-oxidizer and the percentage of vehicles requiring traps. Both the first costs (trap-oxidizer system) and the lifetime costs appear substantial on an absolute basis, while on a relative basis they represent a modest 0.5-2 percent increase. Under the base scenario, 48-70 percent of each vehicle type will experience these costs, while nearly all vehicles will be equipped with traps under the stringent scenario.

While consumers purchasing diesels will experience a 0.5-2.0 percent increase in transportation costs, these costs will likely reduce the demand for diesel-powered vehicles. Table 7 shows estimates of the sales reductions projected under the various scenarios. For LDDVs and LDDTs, the impacts are

Table 6

Total Cost to Consumers of Owning and Operating a Light-Duty Diesel Equipped With a Trap-Oxidizer and the Percentage of Vehicles Requiring Traps (1983 dollars)*

	LDDVs	LDDTs	MDVs	LHDVs	HHDVs
Trap-Oxidizer System:	$185-213	$187-211	$363	$556	$652
Maintenance Costs	$22	$22	$22	$44	$44
Maintenance Savings	($21-36)	($22-36)	($39)	($61)	($97)
Cost of Fuel Economy Penalty	$33-52	$41-55	$126	$386	$917
Total Cost to Consumer:	$219-266	$229-252	$472	$925	$1,516
Total Cost of Owning and Operating Vehicle	$19,418	--	--	--	$274,911
Cost Increase Due to Trap-Oxidizer	1.4-1.5%	--	--	--	0.6%
Vehicles Requiring Traps (%)**					
Base Scenario	47.6	56.2	70	70	70
Stringent Scenario	95.1	94.7	98	98	98

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* All costs are discounted to year of vehicle purchase using a 10 percent discount rate.

** Presumes the availability of averaging for all vehicle classes and stringent 1.0/1.2 g/mi NOx standards for LDDVs and LDDTs, respectively. Also that the current fleet composition is retained in the future.

Table 7

Diesel Sales Reductions*

	1987	1990	1995
Base Scenario			
LDDV	7.4%	5.4%	4.8%
LDDT	10.1%	7.9%	6.7%
MDV**	3.4%	3.4%	3.4%
LHDV	1.5%	1.5%	1.5%
HHDV	0.7%	0.7%	0.7%
Stringent Scenario			
LDDV	11.1%	7.2%	7.9%
LDDT	14.7%	11.6%	9.7%
MDV	4.8%	4.8%	4.8%
LHDV	2.2%	2.2%	2.2%
HHDV	1.0%	1.0%	1.0%

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* Presumes the availability of averaging for all vehicle classes and stringent NOx standards for LDDVs and LDDTs.

** Estimates for MDVs, LHDVs, and HHDVs are based on estimated elasticities which are not time dependent.

substantial: 5-15 percent. However, nearly all these lost diesel sales should be made up by increased sales of gasoline-fueled vehicles. Since all LDDV and LDDT manufacturers also produce gasoline-fueled vehicles, none should necessarily suffer a significant loss of sales.

The question arises as to whether these sales reductions would force any particular manufacturer out of the LDD market entirely. This cannot be answered definitively, given the recent volatility of the LDD market. If the diesel penetrates the market as projected under the best sales estimate used in this report (10 percent for LDDVs, 30 percent for LDDTs), it is doubtful that any manufacturer would be forced out by a 5-15 percent drop in diesel sales. On the other hand, if diesel sales, particularly LDDV sales, did not reach these levels, then such reductions could cause some to leave or never enter the market.

The projected sales reductions for heavy-duty diesels are considerably smaller than those for LDDs, though they are probably more prone to error. The approximate loss of one percent for HHDVs is very small. The losses in the other two subclasses (MDVs and LHDVs) are larger, ranging from about 2-12 percent. However, in these lighter two subclasses, a sizeable portion of the lost diesel sales would again be made up by sales of gasoline-fueled vehicles and the diesel fraction of sales within these two subclasses should still increase markedly. Thus, in general the HDD sales impacts are probably insignificant.

One caution should be made concerning these costs and sales impacts. They presume that the only control technology available to reduce particulate emissions is the trap-oxidizer. As will be seen in Section III, other techniques are available which are less costly, but which cannot provide the same degree of reduction as trap-oxidizers. Thus, the cost of standards less stringent than those of the base scenario may be substantially less than those shown in Tables 6 through 7 and may not have associated with them the concerns indicated above. Also, the use of these less costly techniques will also reduce the costs and economic impacts associated with the base and stringent scenarios as well, particularly the former. This will be considered in greater detail in both Sections III and IV.

C. Cost Effectiveness

The cost-effectiveness analysis of Chapter 9 identified two methods for comparing the cost effectiveness of various controls. Among mobile sources, an urban based cost effectiveness appeared adequate. When both stationary and mobile sources are involved, an air quality discounted cost effectiveness appeared to be more appropriate, though still not entirely adequate.

The results of the mobile source comparison are shown in Table 8. As can be seen, except for MDVs, the cost effectiveness of all the vehicle classes is very similar under both the base and stringent scenarios. Thus, any preference for control based on cost-effectiveness considerations would be very small, and, practically speaking, need not be considered.

One caution should be mentioned. The cost effectiveness of both scenarios for LDDVs and LDDTs is very dependent on the level of their NOx standards. If higher NOx standards were considered (e.g., 1.5 and 1.7 g/mi, respectively), the dollar per ton figures would nearly double due to the lowering of engine-out particulate levels. This aspect will be considered later in Section IV of this chapter as the possibility of higher NOx standards are considered.

The results of the cost-effectiveness comparison between stationary sources and mobile sources contained in Chapter 9 showed diesel particulate control to be cost effective relative to stationary source control when air quality impact was considered. However, other important parameters, such as the ability to focus control in those areas needing it, could not be included and these could significantly affect the outcome of the analysis. At this point, then, all that should be said is that there is no evidence that diesel particulate control should be avoided due to the better cost effectiveness of stationary source control.

III. Identification and Assessment of Viable Control Options

The purpose of this section is to review all available techniques for controlling diesel particulate emissions, formulate specific control options, and assess their relative technological feasibilities and costs (if not already addressed above). This will be done in two parts.

The first part will review conventional techniques available for diesel particulate control (i.e., hardware-oriented techniques applied to vehicles nationwide), first for light duty and second for heavy duty. Discrete levels of control will be identified and their relative feasibility and costs discussed. The second part will present a number of more innovative possibilities for HDD particulate control, including fuel-related techniques and controls oriented toward urban fleets.

Table 8

Urban Based Cost-Effectiveness Relative to the Relaxed Scenario (1983 dollars per metric ton)[1]

	Base Scenario	Stringent Scenario
LDDV	15,400	18,900
LDDT	16,100	17,300
MDV	20,000	20,000
LHDV	11,000	11,000
HHDV	11,000	11,000

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[1] Presumes averaging for all classes were applicable and stringent NOx standards for LDDVs and LDDTs.

The available control techniques for LDDVs and LDDTs are discussed together because of the great similarity of their engine/vehicle designs and their usage patterns. All of the evidence generated over the past 10 years, for both gasoline-fueled and diesel-powered vehicles alike, has demonstrated that LDDVs and LDDTs can meet essentially the same emission standards; the LDDT standards only requiring a small upward adjustment based on their heavier maximum weight.

The same approach is taken toward HDDs, where trucks and buses of vastly different sizes are grouped together for the purpose of discussion. Here again, with the possible exception of trap-oxidizers, the available control technology is applicable across the board to all HDD engines/vehicles. With respect to trap-oxidizers, it may be an easier technological task to apply these devices to the smaller, lighter HDDs than the huge tractor-trailers. However, as indicated in Table 3, even including all HDDs up through Class VI in this "lighter" grouping affects less than 10 percent of total urban diesel emissions. If the control technologies applicable to these two subclasses were more clearly different and the lighter subclass had a more substantial emission reduction potential, then it may be desirable to regulate them separately. However, since this is not the case under the conventional control options, they are treated together below. In the section discussing innovative control options, their potential for separate regulation in examined.

A. Conventional Options

1. Light Duty

The options available for the control of LDD particulate based on hardware-oriented techniques are fairly straightforward, because the available control techniques are few in number. Moving from least stringent to most stringent, there are four possibilities.

One, no control beyond that already applied can be required (i.e., the relaxed scenario). This approach would result in various particulate standards depending on: 1) the level of the NOx standard, and 2) whether averaging is made available with such a relaxed standard. Table 9 presents the corporate average particulate levels for LDDVs and LDDTs under stringent NOx standards (1.0 g/mi and 1.2 g/mi, respectively) and averaging. While fleet average emissions would be 0.42 and 0.52 g/mi, respectively, the particulate standards would have to be higher to accommodate higher than average manufacturers (0.60 g/mi for LDDVs and 0.56 g/mi for LDDTs). Without averaging, the standards would have to be higher still, as shown in Table 9.

Table 9

Individual and Overall Current Corporate Average Particulate Standard Levels for LDDVs and LDDTs Under Option 1 (grams per mile)*

Manufacturer	LDDV	Manufacturer	LDDT
General Motors	.50	Ford	.30
Volkswagen	.21	General Motors	.56
Nissan	.29	Isuzu	.33
Mercedes-Benz	.60	Nissan	.37
Isuzu	.22	Mitsubishi	.43
Audi	.26	Toyota	.20
Peugeot	.36	Volkswagen	.32
Volvo	.41	Toyo Kogyo	.30
Sales-Weighted Industry Wide Average	.42	Sales-Weighted Industry-Wide Average	.52
Resultant Averaging Standard	0.60	Resultant Averaging Standard	0.56
Resultant Non-Averaging Standard	0.63	Resultant Non-Averaging Standard	0.64

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* Presumes stringent NOx standards of 1.0/1.2 g/mi for LDDVs and LDDTs, respectively, and the current fleet composition.

Two, a moderate degree of additional control could be obtained via non-trap technology (i.e., between the relaxed and base scenarios). For example, electronic fuel injection and sophisticated electronic exhaust gas recirculation (EGR) systems (to reduce negative impact of stringent NOx standards) appear to be available and able to provide some control. There is also the possibility that certain high-emitting engine lines may be dropped in this timeframe due to fuel economy and other pressures. Overall, these techniques could significantly reduce particulate emissions from the levels shown in Table 9, depending on the manufacturer and the desired stringency. As determined in Chapter 10, particulate standards under this option would be approximately 0.30 g/mi and 0.35 g/mi for LDDVs and LDDTs, respectively.

Three, the current trap-based standards of 0.20 g/mi and 0.26 g/mi for LDDVs and LDDTs, respectively, are potential candidates (i.e., the base scenario). As indicated above, these levels may be achievable without traps by some or even most manufacturers with extensive use of non-trap technology, even with stringent NOx standards. However, some use of traps would be likely, especially if the higher emitting General Motors Corporation (GM) and Mercedes-Benz models remained in production.

Four, full trap-based standards of 0.08 g/mi and 0.105 g/mi, respectively, comprise the most stringent option (i.e., the stringent scenario). These standards would require traps on essentially all vehicles unless some unforeseen breakthrough occurred in engine-related technology. Non-trap technology would not be as desirable under this option, from the manufacturer's point of view, since in any case the great majority of vehicles would still require traps.

The first option would of course be technologically feasible since no new technology would be required. (The NOx/particulate trade-offs used to convert particulate levels under the current NOx standards to those under the stringent NOx standards assume only the use of current EGR systems and their associated particulate penalties.)

The second option would also be relatively easy to demonstrate as being feasible. As discussed in Chapter 10, sophisticated EGR systems are available and are already being applied on a few 1984 vehicles in California to comply with their 1.0 g/mi NOx standard. These systems should be able to reduce the particulate penalty associated with the stringent NOx standard by one-half. While the effect of electronic injection is uncertain, it should provide a benefit for the highest emitting engines; at least again improving the

NOx/particulate trade-off. Finally, it appears almost certain GM will drop their 5.7-liter engine by 1987 of their own accord. (GM has already dropped this engine from its 1984 California model line and has indicated its intent to discontinue this engine Federally beginning in 1986.) This single change would have a drastic effect on their corporate average emission level and would significantly reduce the number of traps required under this scenario.

The third option is basically an extension of the second, technologically. However, as these standards would probably be lower than that achievable via non-trap technology, the likelihood of traps being necessary would be a distinct possibility, and would need to be addressed. The record for trap-oxidizer feasibility on LDDs, though, is already well established. Thus, this option is feasible.

The fourth option, too, only requires that traps be feasible. However, because highly efficient traps would be required on nearly all vehicles, some additional leadtime beyond 1987 would appear to be necessary. (California is not requiring this level of control until 1989.) However, the basic feasibility of these levels does not appear to be in question.

Concerning the costs of these options, the only cost not identified in Section II was that of the non-trap technology. The cost of electronic fuel injection can be substantial (i.e., $100). However, there are other benefits involved besides emission control, such as improved fuel economy and performance. As some manufacturers are already planning to apply this technology on their engines, it is evident that these non-emissions related concerns justify most if not all of this cost. Thus, this technology should not be costly from the point of view of emission control. In addition, the primary cost of a sophisticated EGR system would be the electronic control unit already included in the cost given above for electronic fuel injection. Thus, this technique should also have a modest net cost. Overall, the portion of hardware costs associated with emission control should be about $25 per diesel vehicle. As such, the urban cost effectiveness for this scenario appears to be very good at about $2,500-$3,500 per metric ton.

2. Heavy Duty

The conventional control options available for HDDs are more complex than those for LDDs. This is due to the fact that the CAA contains strong mandates for the control of both HDD particulate and NOx emissions While the ultimate level of the

HDD NOx standard is specified in the CAA, the technology is not available to achieve that level. Thus, the CAA provides that EPA set stringent interim NOx standards based on criteria similar to those specified for the particulate standards. This is unlike the light-duty situation, where the NOx standard is specified in the CAA.

Despite this difference, generally speaking there are the same four options available for HDD particulate control as were identified for LDDs with the added complexity resulting from a variable NOx standard. These four options are examined below.

The first option is again no, or little, further control (i.e. the relaxed scenario). While current HDD particulate and NOx levels are around 0.7 and 7 grams per brake horsepower-hour (g/BHP-hr), respectively, testimony at a public hearing on this issue identified 0.6 and 6.0 g/BHP-hr as reasonable, very short-term standards that would require a minimal amount of very basic, engine-related control (e.g., injection timing and injector design). Thus, it does not appear necessary to consider higher levels. The 0.6/6.0 g/BHP-hr levels could be shifted somewhat toward lower particulate or lower NOx levels if a strong priority existed for one or the other. However, the 6.0 g/BHP-hr level probably represents the reasonable limit for short-term (i.e., 1987-88) NOx control without severely impacting particulate levels and, likewise, the particulate reduction achievable from raising the NOx standard would not justify the increase in NOx emissions. Thus, these levels represent the practical levels for short-term particulate and NOx standards using readily available technology.

The levels associated with more advanced non-trap HDD control technology (i.e., the second option, between the base and relaxed scenarios) are more difficult to pinpoint than those for light duty because less data are available. Electronically controlled fuel injection, adiabatic engine techniques, and advanced EGR appear to have the most promise. (Oxidation catalysts are probably not feasible here due to the low exhaust temperatures of HDDs.) An estimate of the maximum effectiveness of these concepts is that they could reduce emissions down to particulate and NOx standard levels of 0.4 and 4.0 g/BHP-hr, respectively, giving equal weighting to both particulate and NOx control. It is doubtful if more NOx control could be obtained even allowing some increase in particulate levels, due to engine durability and fuel economy concerns. However, if achieved, such additional NOx control would almost certainly bring with it large increases in particulate emissions, well above 0.6 g/BHP-hr. If the greater weight were given to particulate control, levels around 0.3 g/BHP-hr would probably be achievable. However, NOx levels would almost certainly increase to 5.0-6.0 g/BHP-hr. This would mean that NOx emissions would increase 10-20 metric tons for every metric ton of particulate reduced. Given the present acid rain and ozone problems, this does not appear to be a desirable trade-off. Thus, the simultaneous reduction of both particulate and NOx emissions appears more reasonable than a trade-off of the control of one pollutant for the other, and 0.4 and 4.0 g/BHP-hr, respectively, will be considered the most stringent standard levels associated with the second option.

Analogous to the third LDD option (i.e., the base scenario) is the 0.25 g/BHP-hr standard currently proposed for HDDs in 1986. As implied in the discussion of non-trap technology above, this level, even more than the 1987 LDD standards, is dependent on the use of traps. It does not appear feasible for non-trap controls to even approach this level. However, with the application of traps, the level of the NOx standard becomes more flexible. Traps, in general, are sufficiently efficient to provide compliance with the 0.25 g/BHP-hr standard with engine-out particulate levels of even 1.0 g/BHP-hr. Assuming the availability of averaging, the only effect of additional NOx control would be an increased number of traps which had to be applied. However, fuel economy and durability penalties associated with NOx standards below 4.0 g/BHP-hr would still prevent lower NOx standards from being practical. Thus, 4.0 g/BHP-hr will represent the NOx standard associated with the trap-based standards as well as the Option 2 non-trap standard.

Finally, there is the full, trap-based level (i.e., the stringent scenario). Assuming a starting point of 0.6 g/BHP-hr, a particulate standard of 0.1 g/BHP-hr should be attainable with high-efficiency traps on essentially all engines. The NOx standard associated with this option would again be 4.0 g/BHP-hr, not because of the need for particulate control, but because of fuel economy and durability considerations.

The feasibility of the first option, 0.6 and 6.0 g/BHP-hr for particulate and NOx, respectively, is relatively straightforward and achievable by 1987. This would represent a 2-year delay with respect to the CAA mandate that a 1.7 g/BHP-hr NOx standard (or an alternate standard representing the technologically feasible limit) be implemented by 1985 and a 6-year delay of the CAA mandate that a particulate standard be implemented by 1981. In addition, the 0.6 and 6.0 g/BHP-hr standards would not represent true technology-forcing standards, as indicated by the Act. However, they do represent the feasible limit given the leadtime available. Since the promulgation of such standards cannot take place prior to early

1985, the minimum leadtime of 4-years also mandated by the CAA would not be available. However, this 4-year leadtime was intended by Congress to be associated with a technology-forcing standard, which as indicated above, is not the case here. As outlined in the Agency's position when it implemented the 1984 HC and CO standards for heavy-duty engines, when the model year mandate for implementing a such standard has been missed, implementing the standard at the earliest reasonable date takes precedence over the 4-year leadtime requirement, provided that sufficient leadtime is available, which is the case here. In fact, manufacturers have had notice of even more stringent standards since January of 1981 when the 0.25 g/BHP-hr particulate standard was proposed for the 1986 model year and advance notice was given of a 4.0 g/BHP-hr NOx standard for the same model year.

The demonstration of the feasibility of the second option, 0.4 and 4.0 g/BHP-hr without traps, however, is more difficult and would require leadtime in line with the 4-year requirement of the CAA. None of the three control techniques associated with this option, electronic fuel injection, electronically controlled EGR, or adiabatic combustion techniques, have yet to be commercialized on HDD engines, though the first technique will appear on portions of manufacturers' fleets as early as 1986 and on a majority of HDD engines by 1988. However, implementation of the other two control techniques is further away. Because of durability concerns in the HDD industry, it is sometimes difficult to require basic engine modifications without granting leadtime for necessary durability evaluation. Thus, it would not appear feasible to require the application of such techniques until 1990, which would be the year for a revised NOx standard if the first standard were implemented in 1987. Even with this leadtime, it would not be easy to firmly demonstrate at this time the feasibility of a 33 percent reduction in both NOx and particulate emissions. However, at the same time, it should be difficult for the industry to argue their infeasibility given the mandates in the Act, the years of leadtime available, and the fact that these concepts have been tested on prototype engines and have shown substantial emission reduction potential.

Both the third and fourth options depend on the feasibility of trap-oxidizers. While the basic feasibility of trap-oxidizers for HDDs is not in question, the development of HDD trap technology appears to be significantly behind that for LDDs. There appear to be a number of technical reasons for this. One, the exhaust characteristics of HDDs make it much more difficult to initiate and control regeneration compared to LDDs. HDD exhaust is generally cooler, of much greater volume, and both temperature and volume are subject to wide fluctuations due to the more extreme operating conditions seen by large trucks relative to those of cars. Two, the extremely long life of HDDs also requires a more durable trap system. Three, the importance of fuel economy to commercial operators may be a significant impetus to tamper, since traps will cause some fuel economy penalty (possibly 1-3 percent) due to increased backpressure. This last difficulty is potentially the most important, since it is unlikely that the fuel economy penalty can be removed through further development.

There appears to be a regulatory reason, as well, why HDD trap development is behind that for LDDs. Trap-based LDD particulate standards were proposed in 1979 and promulgated in 1980. These actions communicated a serious regulatory intent to the LDD industry and a commitment to the degree of emission reduction achievable via trap technology. Even the regulatory relief program of 1981 focused on delay and not relaxation. A commitment to reevaluate the need for the standards was made (this study is the fulfillment of that commitment), but it was never communicated that the study had a prejudged outcome (i.e., relaxation). Rather, it was emphasized that the study would be highly technical and objective. The result has been continuous progress in LDD trap development, even though it has been difficult to obtain a clear picture of the manufacturers' progress since they have a strong incentive to withhold such information.

In contrast, EPA did not propose a trap-based HDD particulate standard until 1981, and then almost immediately suspended regulatory activity in this area for a year and a half. After holding a hearing on the proposal in mid-1982, EPA announced it would repropose the particulate standard with the proposal of the HDD NOx standard, which is now scheduled for Summer of 1984. Thus, the Agency has not conveyed a strong regulatory intent, but rather one of accommodation.

As a result, the trap-development programs of the HDD manufacturers with one exception, are years behind those of the LDD manufacturers. For example, two of the five major domestic HDD manufacturers have done little trap testing to date, while two others are still focused on bench testing. Little actual engine or vehicle testing has been done and the focus of the effort is as much to raise problems as to solve them. Only one HDD manufacturer is known to have proceeded actively in trap development, taking full advantage of available LDD experience.

Nonetheless, even though EPA proposed a trap-based partculate standard in 1981 and Congress mandated strong controls for the 1981 model year, it is clear technologically that more time is needed beyond 1987 or even 1988 to implement traps on HDDs. Sufficient leadtime is probably available for 1988 or 1989, at least until a later date when further progress can be assessed. Thus, the standards of either the third or fourth options could probably be implemented as early as the 1989 model year or more certainly for the 1990 model year. The statutory framework for the setting of the NOx standard requires a revision of the standard every three years (until the 1.7 g/BHP-hr level is reached). Thus, if the first option were implemented in 1987, 1990 would be the first possible year for a revised NOx standard and would also allow a reasonable amount of leadtime for trap introduction.

Concerning costs, again as with LDDs, the only cost not yet identified is that associated with the non-trap technology of the second option. Here the costs of this technology are more difficult to identify than they were in the LDD situation, because adiabatic techniques, electronic fuel injection, and EGR could involve fundamental changes to the engine or fuel system and their cost is difficult to identify. Fortunately, an answer may lie in the fact that manufacturers are moving toward implementing these techniques in the late 1980's or early 1990's even without impetus from emission standards. Thus, their cost attributable to emission control should be small, given sufficient leadtime, and they should be cost effective.

B. Innovative Control Options

The search for innovative strategies for the control of HDD particulate emissions was a result of three factors. One, HDD particulate emissions constitute the major portion of total urban diesel particulate emissions. Two, traps do not appear to be feasible for HDDs prior to 1989 and even after that in-use tampering may be a problem. Three, less than 30 percent of all HDD particulate is emitted in urban areas, although this minority of HDD particulate emissions still represents more than half of all urban diesel particulate emissions. The logical conclusion is that: 1) major reductions in a sizable portion of the urban diesel particulate emission inventory will not be achievable in the near term, and 2) even in the long term, the traditional strategy of controlling all vehicles sold nationwide will, in a sense, be somewhat inefficient. This led to a search for strategies for controlling HDD particulate emissions specifically in urban areas only.

Two strategies appear to have merit. One would selectively regulate those types of HDDVs which are heavily urban-oriented (an urban vehicle option), applying lesser control to those which are rural-oriented. The other would selectively regulate urban fleets (an urban fleet option); it being easier to monitor maintenance with fleets and fuel modification becoming a possibility due to their captive fueling capability.

The selective regulation of urban-oriented HDDVs would require the differentiation of vehicle types based on objective criteria that would not be easily modifiable (i.e., weight, number of axles, length, general description of load carrying portion, etc.). Two or three vehicle types come to mind as heavily urban oriented: transit buses, garbage trucks, and cement mixers. An analysis of their use nationwide shows that, if one assumed their use was entirely urban, then their vehicle miles travelled (VMT) would represent roughly 20 percent of all HDD urban VMT, with transit buses representing slightly more than half of this figure. At the same time, the great majority of the remaining HDD VMT nationwide is associated with very general types of vehicles, such as panel trucks and flatbeds, which would definitely be used in both urban and rural areas. To affect these vehicles, weight classes could be used. One option would be to include all HDDs in Class VI or below. Such vehicles would be expected to expend roughly half of their VMT in urban areas, so their control would be as cost effective from an urban/rural point of view as that of LDDs. Their inclusion would increase the percentage of affected HDD urban VMT by 20-30 percent, to a total of 40-50 percent.

In addition, there are certain cities where bus usage alone represents a sizeable majority of HDD VMT. For example, bus VMT represented roughly 60 percent of all 1980 HDD VMT in New York City and Chicago. (In contrast, the same figure for Los Angeles was 4 percent.) However, the dieselization of the smaller HDV classes will reduce these fractions in future years. Again for example, bus VMT is projected to represent 33 and 42 percent of 2000 HDD VMT in New York City and Chicago, respectively, a decrease of one-third to one-half from current levels, but still significant fractions. An additional reason to focus additional control on transit buses is the fact that they are publicly operated and are heavily subsidized by the Federal government. It would appear to be particularly appropriate for these buses, which serve the public interest, to demonstrate the ability of technology to control the very apparent particulate emissions of diesel engines. Thus, there is some potential for a large degree of additional control in particular cities, and for a significant degree of control in other cities via an urban vehicle option, as well as a certain appropriateness to focus control on transit buses.

The second strategy, instead of identifying vehicles by functional design, focuses on user type and location (i.e., urban commerical and public fleets). This strategy has the advantage of affecting nearly all urban-oriented diesels. The difficulty arises in defining an urban fleet and enforcing the requirement. A fleet may be located in an urban area but involve almost entirely over-the-road operation (e.g., most tractor-trailer fleets are located in suburbs). Thus, weight limitations or other means of distinguishing certain trucks may need to be added to address this problem. Also, there is a reliance on the HDD purchaser buying the proper vehicle from the dealer. Since most dealers sell to both urban and non-urban users, deception or conscious manipulation would be a definite possibility. Finally, this strategy could create enough incentive to cause a firm to relocate.

Despite these problems, this strategy should have the ability to address the large majority of urban HDD emissions. While figures are not readily available on the relative mileage of fleets versus total vehicle usage, the three vehicle types mentioned above for the first strategy would definitely be included plus even larger utility, city, and delivery fleets. Thus, a sizeable majority of urban HDDs should be affected.

Both of these urban strategies would also have a second advantage over the conventional nationwide control approach. While selective regulation could be oriented toward requiring more stringent emission controls to be part of the engine or vehicle, it could also be oriented toward requiring a cleaner fuel to be used, since urban fleets usually have their own fueling facilities. Gasoline, for example, would certainly be cleaner with respect to particulate emissions. However, a large fuel economy penalty would be involved and emissions of lead would increase, at least until these vehicles are equipped with catalysts. A better alternative would appear to be methanol. Methanol engines have little or no particulate emissions, low NOx emissions and can have fuel efficiencies approaching those of the diesel. And, while methanol is currently not available at service stations, it is readily available in bulk quantities and could easily be bought and used by fleets with their own facilities.

Engine manufacturers could certainly produce a methanol engine if there was a market for them. A German manufacturer, M.A.N., already has a production-ready methanol-fueled diesel engine and Detroit Diesel has recently developed a methanol-fueled diesel bus engine under contract to the California Energy Commission for fleet testing in San Francisco. Given the current and projected future glut in the methanol market (100,000 to 200,000 barrels per day), fuel supply should also not be a problem. Roughly 67,000 barrels a day of methanol would be required to fuel every transit bus in the U.S. Since conversion to methanol would only occur through the purchase of new buses, it would take a number of years for even half of the bus fleet to convert and the present methanol supply should be more than sufficient.

The question is simply one of cost, primarily of the fuel. A methanol engine should cost approximately the same as a diesel, and its fuel efficiency should also be roughly the same. However, methanol is likely to be more expensive on an energy and mileage basis. Overall, then, a switch to methanol would likely increase operating costs, at least as compared to current petroleum prices.

To estimate this cost, estimates of the prices of both diesel fuel and methanol are needed. No. 1 diesel fuel currently costs roughly $0.90 per gallon delivered to a transit authority, without tax. Methanol is currently available for $0.45-0.50 per gallon on the gulf coast. Both of these prices are depressed because of the depressed worldwide economy, but methanol the more so, its price being 30-35 percent below that of two years ago. Even though a methanol production surplus is projected through the rest of this decade, it should not be as severe as the current situation and the price will likely increase in terms of 1983 dollares. Due to the uncertainty in the degree of this increase, a range of $0.50-0.70 per gallon, including bulk distribution, will be used.

Transit buses obtain a fuel economy of about 5 miles per gallon on No. 1 diesel fuel. Since methanol contains only 44.5 percent of the energy per gallon of No. 1 diesel fuel, the fuel economy of a methanol-fueled bus would only be 2.2 miles per gallon. Assuming the relative fuel prices remain steady in the future, the fuel costs of a diesel bus would be roughly $0.18 per mile and that of a methanol bus would be $0.22-$0.31 per mile, or 25-75 percent higher. At 30,000 miles per year, this translates into an annual cost of using methanol of $1,200-$3,900 per year, or a net present value lifetime cost (15 years) of $9,100-$29,700 in the year of vehicle purchase. With fuel costs being 8 percent of overall costs, a switch to methanol would increase the overall cost of operating a bus by 2-6 percent.

However, it is important to consider that a methanol-fueled bus engine would meet any particulate and NOx standards conceivable for diesel engines without any additional hardware or adjustments. Thus, the cost of bringing a diesel engine to these levels must be considered. For particulate control, a trap would certainly be necessary at a cost of $600-700 initially, and a 2-percent fuel economy penalty. For NOx control, severe timing retard or an exhaust gas recirculation system would be needed, costing $0-200 initially and a 5-10 percent fuel economy penalty, and yet still not achieve the level of the methanol engine. Overall, these control costs would increase the overall costs of owning and operating a diesel bus by $460-770 per year, or $3,500-5,800 over the life of the bus (net present value in year of vehicle purchase). Thus, it would appear that the methanol option would still be more costly, though it should be more easily achievable technically and provide greater NOx control.

One final point bears heavily on the relative economics of using methanol versus diesel fuel. If the price of diesel fuel rises more rapidly in the future than that of methanol, which is possible, at some point methanol will be less costly to use. However, predicting future petroleum prices is beyond the scope of this study.

One method to at least partially avoid the issue of cost would be to implement a particulate standard for urban vehicles which could be met with either trap-oxidizer technology or methanol. Methanol engines could easily be feasible by or even before the feasibility of traps. Thus, engine/vehicle manufacturers would have a choice as to how to meet the standards, via traps or methanol, and the overall approach would involve less risk than if either one were the only potential option. Even if traps are eventually used, this strategy would be 3 to 4 times as cost effective as a nationwide trap-based strategy.

Of the two options, the urban-vehicle (buses, garbage trucks, cement mixers) strategy appears to be preferable, primarily because it avoids the implementation and enforcement problems of the urban-fleet strategy. The cost issue still needs to be addressed in further depth. However, the potential for significant, cost-effective emission reductions is present. For example, either trap technology or methanol could provide compliance with a 0.25 g/BHP-hr standard. Nationwide, assuming such engines just comply with the 0.25 g/BHP-hr standard, this strategy could reduce 1995 HDD urban emissions from 55,000 metric tons per year to 41,000 metric tons per year, or by about 20 percent, compared to a 0.40 g/BHP-hr standard for all HDDs. Even greater reductions would be achieved if methanol engines were chosen as the compliance strategy, since these engines should emit well below 0.25 g/BHP-hr particulate. Thus, this approach merits further consideration.

IV. Discussion of Control Options

The previous section laid out the available control options for both LDDs and HDDs, including two innovative control strategies for HDDs. In this section, these options will be compared and evaluated and, where possible, conclusions drawn. As in the previous section, LDD control options will be discussed first, followed by those for HDDs, as the analyses in each area are for the most part independent.

A. Light Duty

The summary of environmental impacts in Section II demonstrated a need for further control beyond the relaxed scenario. It also showed that LDDs were roughly half the problem (i.e., contributed half of the emissions). A need to control LDD emissions beyond current levels was evident.

The analysis in Section II identified three further levels of control beyond current emission levels; speaking in terms of LDDV standards, 0.25-0.3 g/mi, 0.2 g/mi, and 0.08 g/mi. All three standards are demonstrably feasible, though the 0.08 g/mi level would probably require some time beyond 1987 before implementation, even with the availability of averaging.

At the same time, the summary of costs in Section II showed potentially significant economic impacts for the trap-based standards if the trap-oxidizer were the only control technique employed. While both the initial and lifetime cost impacts are only about 2 percent of vehicle purchase cost and total lifetime cost, this small impact can have a sizeable effect on sales of a consumer alternative like LDDs, since their desirability is very sensitive to small relative economic changes. As was seen, equipping all vehicles with traps could reduce LDD sales by 7-15 percent. This is a sizeable reduction even considering the fact that most of these lost sales would be made up by sales of gasoline-fueled engines.

Table 10 summarizes the urban emissions, various cost information, and urban cost effectiveness for the four LDD control options. The values presented in this table differ somewhat from those shown in Section II primarily because the voluntary elimination of GM's 5.7-liter diesel engine is assumed here. Urban emissions are shown in Table 10 as an indicator of environmental effects since all of the health and welfare impacts discussed in Section II are essentially proportional to urban emissions.

As can be seen, there is a fairly steady and consistent change in emissions from option-to-option. The cost and sales impacts trends are also reasonably steady. The second option should have a fairly low cost due to the other benefits associated with the use of electronics (e.g., improved fuel economy, electronic capability for peripherals, etc.). For option 3 and 4, the use of traps is more costly and hence, more liable to affect sales (Options 3 and 4). These affects are more pronounced under the fourth option. Manufacturers would definitely use traps on nearly all vehicles because the sizeable reductions required could not even be approached with non-trap technology.

Table 10

Summary of Significant Information for the LDD Control Options*

LDDV/LDDT Standard Options (g/mi)	1995 Urban Emissions (m. tons)	LDDVs/LDDTs Requiring Traps (%)	Average Lifetime Cost Per Vehicle (1983 $)	Sales Impact (%)	Incremental Urban Cost Effectiveness ($/m. ton)
1. 0.6/0.56	58,600 (0%) [**]	0	0	0	-
2. 0.3/0.35	43,400 (30%)	3-7	15-30	0-1	2,500-3,500
3. 0.2/0.26***	31,400 (46%)	43	55-85	1-5	8,600-$11,000
4. 0.08/0.105	16,600 (72%)	100	$220-270	7-15	$15,000-17,000

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* Values assume elimination of GM 5.7-liter diesel LDDV.

** Incremental percent reduction from total 1995 urban emissions for LDDs under the relaxed scenario.

*** For comparison, Option 3 with 1.5/1.7 LDD NOx standards would be 1995 urban emissions of: 1) 30,600 metric tons; 2) a 9 percent of LDDs requiring traps; 3) average lifetime costs of $20-$45 per vehicle; 4) a 1-3 percent impact on sales; and 5) an urban cost effectiveness of $2,500-$3,000 per metric ton.

Given that further control is needed from LDDs and that control down to the levels associated with the second or possibly third option appear to be quite feasible and relatively inexpensive, control to this level appears reasonable. However, given the low level of urban emissions under the Option 3 standards and the sizeable costs (direct and indirect) associated with Option 4, it does not appear reasonable to consider the full trap-based standards at this time for LDDs. However, in the long term, the Option 4 standards may be a reasonable control strategy, particularly if LDD sales increase well beyond current projections. Also, the issue of requiring traps on nearly all LDD models will be much clearer in a few years, technologically and economically. The question, then, is whether to implement the Option 2 or the Option 3 LDD standards.

At issue technically is only the number of traps which would be required, since at least one manufacturer is likely to require traps under either option. Standards of 0.3/0.35 g/mi for LDDVs and LDDTs, respectively, would probably eliminate the need for traps for all but one manufacturer. Standards of 0.2/0.26 g/mi would likely require several manufacturers to apply traps on at least some of their vehicles, unless major engine modifications were employed. However, the majority of the incremental urban emission reduction of Option 3, 12,000 metric tons per year in 1995, would not come from the use of trap-oxidizers, but from non-trap techniques and, thus, should be more cost effective than the sole use of traps. This incremental emission reduction is nearly 10 percent of 1995 urban emissions under the relaxed scenario and is larger than total 1995 urban emissions from MDVs and LHDVs combined. At the same time, Option 3 would require the development of trap technology by most manufacturers for application on only a minority of their fleet. Thus, based on costs and emission reductions, there are pros and cons to either Option 2 or Option 3 and either option could be justified and implemented via rulemaking (though retention of the Option 3 standards would of course not require any action on the part of the Agency). However, there are a couple of additional factors which should be considered.

There are two external factors that could affect the LDD situation: the level of the LDD NOx standards and the level of LDD sales. Considering the effect of NOx standards first, the alternatives to the 1.0/1.2 g/mi standards discussed above are all numerically higher: 1.5/1.7 g/mi and 2.0/2.3 g/mi. Chapter 10 showed that the effect on particulate emissions of moving from 1.0/1.2 g/mi to 1.5/1.7 g/mi was much larger than the second step to 2.0/2.3 g/mi. Thus, the discussion here will emphasize the 1.5/1.7 g/mi option.

Table 11 depicts manufacturers' corporate emission averages under the more relaxed 1.5/1.7 g/mi standards. As can be seen, the great majority of both LDDV and LDDT manufacturers are already well within the range of the Option 2 standards, even without considering the application of additional non-trap technology. This reduction in engine-out particulate emissions across the board has the effect of reducing the number of traps required at any standard level. For Option 2, trap usage was shown in Chapter 10 to drop from three to seven percent to about one to three percent, thereby reducing the cost and improving the cost effectiveness of attaining the essentially non-trap standards. At the same time, the reduction in engine-out particulate emissions reduces the emission reduction associated with placing a trap on any individual vehicle. This causes the cost effectiveness of applying trap-oxidizers to worsen by roughly a factor of two under Option 3 compared to that under stringent NOx standards. Thus, the desirability of requiring traps significantly lessens. In general, stringent non-trap standards should be implemented under the 1.5/1.7 g/mi NOx option. This would mean particulate standards in the range of 0.2 g/mi for LDDVs and 0.3 g/mi for LDDTs.

Under 2.0/2.3 g/mi NOx standards, the same arguments are simply carried a step further. Particulate levels are reduced even further than those indicated in Table 11, making even stringent Option 2 standards achievable by nearly every manufacturer without any additional control, trap or non-trap. Option 3 standards should be achievable by nearly all manufacturers with non-trap technology. At the same time, the cost effectieveness of applying traps worsens even further. Here again, the choice should be to implement stringent non-trap standards. In this case, this would mean particulate standards slightly below 0.2 g/mi and 0.3 g/mi, respectively.

Concerning the LDD sales effect, either of two possibilities could occur: 1) sales could be substantially lower than projected by the best estimate, or 2) sales could be substantially higher than projected by the best estimate. If sales did not increase beyond their current levels, the overall need for LDD control would lessen, but not be eliminated since future LDD particulate emission inventories would continue to grow beyond today's levels as shown in Chapter 10. Furthermore, the available technology, cost per vehicle, and cost effectiveness of control would all remain roughly the same, since they are unaffected by sales. Thus, overall, the need for the additional control provided by Option 3 would lessen.

Table 11

Current Individual and Overall Corporate Average Particulate Standard Levels for LDDVs and LDDTs Under Option 1 Presuming NOx Standards of 1.5 g/mi and 1.7 g/mi, Respectively (grams per mile)

Manufacturer	LDDV	Manufacturer	LDDT
General Motors	.16*	Ford	.29
Volkswagen	.20	General Motors	.34
Nissan	.26	Isuzu	.25
Mercedes-Benz	.42	Nissan	.35
Isuzu	.20	Mitsubishi	.39
Audi	.20	Toyota	.19
Peugeot	.26	Volkswagen	.31
Volvo	.29	Toyo Kogyo	.29
Sales-Weighted Industry Wide Average	.20	Sales-Weighted Industry-Wide Average	.33
Resultant Averaging Standard	0.42	Resultant Averaging Standard	0.39
Resultant Non-Averaging Standard	0.43	Resultant Non-Averaging Standard	0.39

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* Assumes the voluntary elimination of the GM 5.7-liter LDDV.

If sales are higher than projected, there is no doubt that the Option 3 standards would be appropriate. As noted in Chapter 2, under worst case sales, 1995 LDD emissions under the Option 3 standards (base scenario) are still at least 40 percent above those shown in Table 11 for the Option 2 standards under best estimate sales indicating the need for the additional control provided by Option 3.

It is conceivable that the level of the LDD standards could be directly tied to the level of LDD sales, if it were decided to relax the existing Option 3 standards to those of Option 2 at this time. Ideally, the sales or sales fraction limit would be on a fleet-wide basis, since it is the fleet-wide diesel penetration that affects emissions. However, in this case, most manufacturers could find the level of the standard changing due to the actions of one or two other manufacturers who decided to invest heavily in diesels. While the more stringent standard itself would be equitable, since it would be the same for all manufacturers, the latter manufacturer(s) would have more advance notice of the switch since it is fully aware of its own future product plans. There would also be little incentive to keep LDD sales low since the level of the standard would only be indirectly connected.

To mitigate this problem, sufficient leadtime would have to be given between the time diesel sales exceeded the limit and the time the revised standard was imposed. While the effort and time needed to reimplement the Option 3 standards would be avoided, this does not appear sufficient to offset the problem that a given manufacturer would not have direct control over whether or not it had to meet the more stringent particulate standard. Direct control would be critical to any sales-based approach since it provides the incentive to keep diesel sales low, as currently predicted by the manufacturers.

The other alternative would be to apply a diesel sales limit to each manufacturer. This would eliminate the need for leadtime since a manufacturer has control over its own sales. A problem arises, however, in how to equitably set the sales limit.

Limits based on absolute diesel sales are probably unacceptable. The wide range of total sales volumes among manufacturers would make this type of limit very inequitable.

A single diesel sales fraction limit is also not likely to be acceptable, since LDD manufacturers currently sell between 1 and 70 percent diesels. A limit near the upper end of this range would allow an enormous increase in fleet-wide diesel sales before most manufacturers would exceed their limit. A limit near the lower end of this range would immediately impose the more stringent standards on a number of manufacturers and issues of equity would quickly be raised. This very issue was raised in 1979 when GM proposed an emissions averaging approach that included both gasoline and diesel vehicles. This approach resulted in increasingly more stringent diesel particulate standards as a manufacturer's diesel sales fraction increased and was immediately challenged by Volkswagen and Mercedes-Benz, manufacturers with high diesel sales fractions.

A diesel sales fraction limit that varied between manufacturers would appear to avoid this problem. However, the question of how to set each manufacturer's limit arises. One approach that appears to minimize inequities between manufacturers would be to set each manufacturer's limit at its maximum diesel sales fraction in recent years plus some margin for moderate growth. Specifically, the model years 1978-82 should include all manufacturers peak diesel fractions and an absolute cushion of 5 percent* should provide for both adequate diesel growth based on manufacturers' current statements and adequate protection for the environment since LDD sales would be held to less than 10 percent of total light-duty sales. Any manufacturer desiring to produce more diesels would have to comply with the existing Option 3 standards, which would be reasonable since manufacturers are currently requesting the relaxation of these standards due to their low projected diesel sales.

In summary, the pros and cons of retaining the current Option 3 standards versus relaxing them to the Option 2 standards are roughly balanced. If Option 3 were chosen, then no action would have to be taken by the Agency. If Option 2 were chosen, it may be appropriate to tie the relaxation to LDD sales levels since manufacturers are currently basing their arguments for relaxation on very low LDD sales. In this case, the best approach appears to be relax the LDD particulate standards to those of Option 2 on a manufacturer-specific basis (all should initially experience the relaxation) as long as that manufacturer maintains its LDD sales fraction at or below its historic (1978-82) peak plus 5.0 percent. Otherwise, it appears appropriate to retain the current Option 3 standards, as it appears inappropriate to grant a permanent relaxation on such volatile grounds as projected LDD penetration.

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* For example, if manufacturers A and B have peak LDD sales fractions of 5 and 50 percent, respectively, their limits would be 10 and 55 percent, respectively.

B. Heavy Duty

The environmental impacts summarized in Section II also demonstrated the need to control HDD emissions beyond the 0.6 g/BHP-hr level. However, unlike the options for further control for LDDs, all three levels of additional control for HDDs could not be implemented in 1987. This occurs because: 1) engine modifications need to be extensively tested for durability due to market demands, and 2) the application of traps on HDDs is more difficult than that on LDDs. Thus, the technology-forcing mandate of Section 202(a)(3)(A)(iii) of the Act notwithstanding, the only option truly viable for HDDs prior to 1990 is the first option.

This first option, 0.6/6.0 g/BHP-hr for particulate and NOx, respectively, is achievable as early as 1987. This level is supported by at least one manufacturer's comments to the proposed 1986 HDD particulate standard as well as by an analysis of current HDD emission levels and projections of minor control techniques applicable in this timeframe.

Given the 3-year standard revision cycle provided by the CAA, the earliest year for a revised set of standards would be 1990. Here, Options 2, 3, or 4 are potentially available. Given the demonstrated need to control HDD emissions and the apparent cost effectiveness of the second option, control to at least this level appears reasonable. The only questions associated with Option 2 are the actual feasible levels of the standards. Standards of 0.4 and 4.0 g/BHP-hr for particulate and NOx, respectively, probably represent the maximum degree of control achievable by 1990 without traps. Thus, promulgation of these standards would represent a commitment to technology-forcing by the Agency, with a potential need for a delay at a later date. The alternative would be less stringent non-trap standards, such as 0.5 and 5.0 g/BHP-hr, respectively, which would provide little additional control over the 1987 standards.

A disadvangage of Option 2 is its effect on future HDD trap development. As alluded to earlier, there is strong evidence that significant development of a new control technology only occurs in response to the promulgation of a standard requiring its use. Thus, if a trap-based standard is not proposed and promulgated for 1990, HDD trap development will not progress and will appear no more implementable in 1990 than it does now.

On the other hand, the near-term promulgation of a trap-based HDD standard for 1990 would also represent a firm commitment by EPA to technology forcing. The problems facing

HDD trap implementation are sufficiently significant at the present time, as were those facing LDD trap implementation in 1979, that there is a fair probability that the standard may not be feasible by 1990 and may require some delay at a later date, and a continued commitment to its eventual implementation. Thus, compliance with this option may be no more difficult than compliance with the second option, though the technical issues involved would be very different (engine related vs. trap related). However, the trap-based option would, or course, provide more control than the non-trap approach, so it has this advantage over Option 2.

With respect to the level of the trap-based standard, 0.25 g/BHP-hr would appear most appropriate. The requirement that essentially all HDDs be equipped with high-efficiency trap-oxidizers via a 0.1 g/BHP-hr standard would be difficult to justify for 1990. The NOx standard would still be 4.0 g/BHP-hr as this level appears to represent the limit of technology regardless of whether traps are applied for particulate control.

The option of implementing an urban vehicle strategy is also available in the 1990 timeframe. There are still some questions surrounding its cost, particularly for buses since the economic viability of transit authorities could be seriously weakened by any additional costs. However, these costs could be mitigated by a Government subsidy for methanol use by transit authorities which would equate the cost of methanol and diesel fuel on an energy basis. This would be consistent with existing government support of the basic availability of urban transit, only adding support for emission control as well. Private fleets should be much better able to absorb such costs, particularly since many such fleets will be switching to diesels from gasoline engines. They will have the option to stay with the gasoline engine, in addition to the options of paying for the additional cost of a trap-oxidizer methanol engine; the methanol option only being viable for those operators with central fuel depots.

Technical feasibility would not be an issue for those vehicles for which methanol is a viable option, since methanol engines should be easily producable. However, for those other vehicles, traps would be required. Given that only Class VI and lighter vehicles would be involved, trap feasibiliy should be more easily demonstratable than with respect to all HDDs, since LDDT trap experience should be more applicable.

Overall, then, there appears to be only one option for 1987, a 0.6 g/BHP-hr particulate standard and a 6.0 g/BHP-hr NOx standard. Three options appear viable for 1990. One, particulate and NOx standards of 0.4 and 4.0 g/BHP-hr, respectively, for all HDDs. Two, standards of 0.25 and 4.0 g/BHP-hr, respectively, for all HDDs. Three, standards of 0.25 and 4.0 g/BHP-hr, respectively, for Class VI and lighter HDDs and transit buses (and possibly garbage trucks and cement mixers) and standards of 0.4 and 4.0 g/BHP-hr, respectively, for all other HDDs. With the third option, the emissions of methanol engines would be included in the averaging process.

References

1. "Regulatory Analysis of the Light-Duty Diesel Particulate Regulations for 1982 and Later Model Year Light-Duty Diesel Vehicles," U.S. EPA, Office of Mobile Sources, February 20, 1980.

2. "Draft Regulatory Analysis: Heavy-duty Diesel Particulate Regulations," U.S. EPA, Office of Mobile Sources, December 23, 1980.

3. "Health, Soiling, and Visibility Benefits of Alternative Mobile Source Diesel Particulate Standards," Final Report, EPA Contract No. 68-01-6596, Mathtech, Inc., Princeton, NJ, December 1983.

Part II

Supporting Technical Analyses

CHAPTER 1

TECHNOLOGY

I. Introduction

The major reductions in diesel particulate emissions available from engine modifications have already been achieved, with the possible exception of electronic control of the fuel injection system. Further major reductions will need to be accomplished through the use of trap-oxidizer systems.

Under the current light-duty diesel vehicle (LDDV) and light-duty diesel truck (LDDT) particulate standards of 0.6 gram per mile (g/mi), no traps are necessary. Since heavy-duty diesels (HDDs) are not currently subject to a particulate standard, traps are not found on current HDDs either. However, the more stringent particulate standards of the base scenario will require traps on many diesels. This chapter investigates each manufacturer's need for trap-oxidizer systems under the LDDV, LDDT, and HDD particulate standards of the base scenario, as well as determining the non-trap particulate emission levels which would occur under less stringent particulate standards of the relaxed (non-trap) scenario.

This chapter is divided into three sections, each in turn addressing LDDVs, LDDTs and HDDs. The section addressing LDDVs is the most detailed, as the methodology for all three sections is therein described. The latter sections only reference this methodology.

The LDDV section itself consists of five parts. The first simply describes the source of the engine-out LDDV particulate levels used in the analysis. The second addresses the NOx/particulate trade-off issue and establishes NOx/particulate relationships to be used in adjusting particulate emission levels to varying NOx levels. While these relationships will have only a limited use here in addressing the base scenario--most LDDVs are at NOx levels near those appropriate to comply with the base scenario's 1.5 g/mi NOx standard--they will be of significant use in addressing the sensitivity of the results of this chapter to varying LDDV and LDDT NOx standards (see Chapter 11). The third part of the LDDV section estimates the equivalent "standard" levels that each LDDV engine configuration could meet without traps and the fourth part converts these engine configuration levels into corporate average non-trap standards achievable by each manufacturer (the relaxed scenario). The fifth and final part will then determine the percentage of LDDVs which will require traps under the base scenario (0.2 g/mi particulate and 1.5 g/mi NOx standards with corporate averaging).

II. Light-Duty Diesel Vehicles

A. Current Levels of NOx and Particulate Emissions

The most convenient and accurate source of current LDDV engine-out particulate levels is the new-vehicle certification program. The first model year in which LDDV manufacturers had to certify to the current 0.6 gram per mile (g/mi) particulate standard was 1982. However, some manufacturers chose to test for particulates in the 1981 model year and then carryover the results for the 1982 model year thereby spreading out the new emissions testing program over two years. Thus, certification test results for LDDV particulate are primarily available for the last two years (ie., the 1982 and 1983 model years), with some data being available from the 1981 model year as well.

All of the 1983 model year LDDV engine families were subdivided into configurations on the basis of transmission type and inertia weight class. The available 1981-83 NOx and particulate test data were then obtained for each of these configurations. These data included emission tests plus fuel economy tests during which emissions were also measured. Both manufacturer tests and EPA tests were included.

A review of the test results of configurations for which testing had been done for both the 1982 and 1983 model years did not show a clear pattern of change from one year to the next, although there was a modest trend for both NOx and particulate to improve with the more recent data. Therefore, it was concluded that only the most recent (1983) test results should be used here when available. However, in the cases where 1983 engine configurations were carried over from 1982 and no 1983 data were available, the 1982 model year certification test results were used.

These most recent test results for each configuration were then examined and outliers excluded before determining the mean for each configuration. In general, outliers were test results greater than 140 percent or less than 60 percent of the mean of the rest of the test results for that configuration. This range may have been somewhat greater or smaller depending on the observed spread and total number of tests. The remaining test results for each configuration were averaged and the resultant means used as the current level of NOx and particulate emissions for each configuration. These engine-configuration means are shown in Table 1-1.

B. The NOx/Particulate Tradeoff

Having established the current NOx and particulate emission levels, an estimate of how the particulate emission level would change if the NOx emission level were increased or decreased was made. Such an analysis was primarily necessary

Table 1-1

Actual, Certification LDDV Particulate and NOx Emission Levels

Manufacturer	Engine Family	Trans.	Inertia Weight Class (lb)	Displacement (liters)	Particulate LMT (g/mi)	NOx LMT (g/mi)
General Motors	Z90	M5	2,500	1.8	.17	1.11
	Z90	L3	2,500	1.8	.13	1.01
	ZK7	L3	3,000	4.3	.22	1.04
	ZK7	L3	3,500	4.3	.25	1.10
	ZT8	L3	3,500	4.3	.21	1.23
	ZT7	L3	4,000	5.7	.32	1.21
	ZT7	L3	4,500	5.7	.37	1.14
	ZT7	L4	4,000	5.7	.37	1.11
	ZT7	L4	4,500	5.7	.40	1.18
Volkswagen	AAO	M4	2,250	1.6	.16	.90
	AAO	M5	2,250	1.6	.19	1.02
	AAO	A3	2,250	1.6	.18	1.01
	JAO	M5	2,500	1.6	.19	1.02
	JAO	A3	2,500	1.6	.17	1.10
	AZ8	M5	2,250	1.6	.22	1.12
	AZ8	M5	2,500	1.6	.20	1.10
	AZ8	A3	2,500	1.6	.29	1.14
	RA5	S4	2,250	1.6	.18	1.02
	BZX	A3	2,750	1.6	.18	1.22
	BZX	M5	2,750	1.6	.22	1.19

Table 1-1 (cont'd)

Actual, Certification LDDV Particulate and NOx Emission Levels

Manufacturer	Engine Family	Trans.	Inertia Weight Class (lb)	Displacement (liters)	Particulate LMT (g/mi)	NOx LMT (g/mi)
Nissan	AF8	M4	2,250	1.7	.17	.82
	AF8	M5	2,250	1.7	.20	.94
	AF8	M5	2,500	1.7	.23	1.00
	AF8	A3	2,250	1.7	.24	.89
	AF8	A3	2,500	1.7	.23	.92
	AFO	M5	3,500	2.8	.22	1.16
	AFO	L4	3,500	2.8	.24	1.32
Mercedes-Benz	501	M4	3,500	2.4	.42	1.11
	501	A4	3,500	2.4	.38	1.15
	508	A4	4,000	3.0	.43	1.26
Isuzu	CD7	M4	2,500	1.8	.19	1.09
	CD7	M5	2,500	1.8	.17	1.21
	CD7	M5	2,750	1.8	.18	1.17
	CD7	A3	2,750	1.8	.16	1.29
Audi	BZ7	M5	2,750	1.6	.22	1.19
	BZ7	A3	2,750	1.6	.17	1.21
	CZ3	A3	3,000	2.0	.19	1.23
	BAX	M5	2,500	1.6	.21	1.08
Peugeot	AA1	M5	3,500	2.3	.28	1.04
	AA1	A3	3,500	2.3	.30	1.01
	BA3	M4	3,500	2.3	.32	.87
	BA3	A3	3,500	2.3	.40	.98
Volvo	AY2	M4	3,500	2.4	.29	1.37
	AY2	A3	3,500	2.4	.27	1.31
	TB0	M5	3,500	2.4	.29	1.17
	TB0	A3	3,500	2.4	.23	1.19

so that the particulate emission level of each configuration under the various NOx standards being considered in the sensitivity analysis could be estimated. However, it is also useful here, since many engine configurations are emitting NOx well below the levels required by a 1.5 g/mi standard and some adjustment of their particulate levels would appear appropriate.

Assuming that only injection timing retard or EGR is used, the general shape of a NOx/particulate tradeoff curve is known to be (NOx emissions in the x dimension and particulate emissions in the y dimension): 1) negative in slope at all points, 2) steeply sloped at low NOx levels, and 3) gently sloped to flat at high NOx levels. Furthermore, it is generally known that the curve shifts outward (ie., upwards and to the right) with increasing engine displacement. Figure 1-1 shows generalized NOx/particulate tradeoff curves and illustrates this shifting effect of engine displacement. Ideally, the specific tradeoff curve would be known for each engine family/configuration. However, such curves are not available. Therefore, an approximate method was developed for predicting particulate emission levels from known NOx levels.

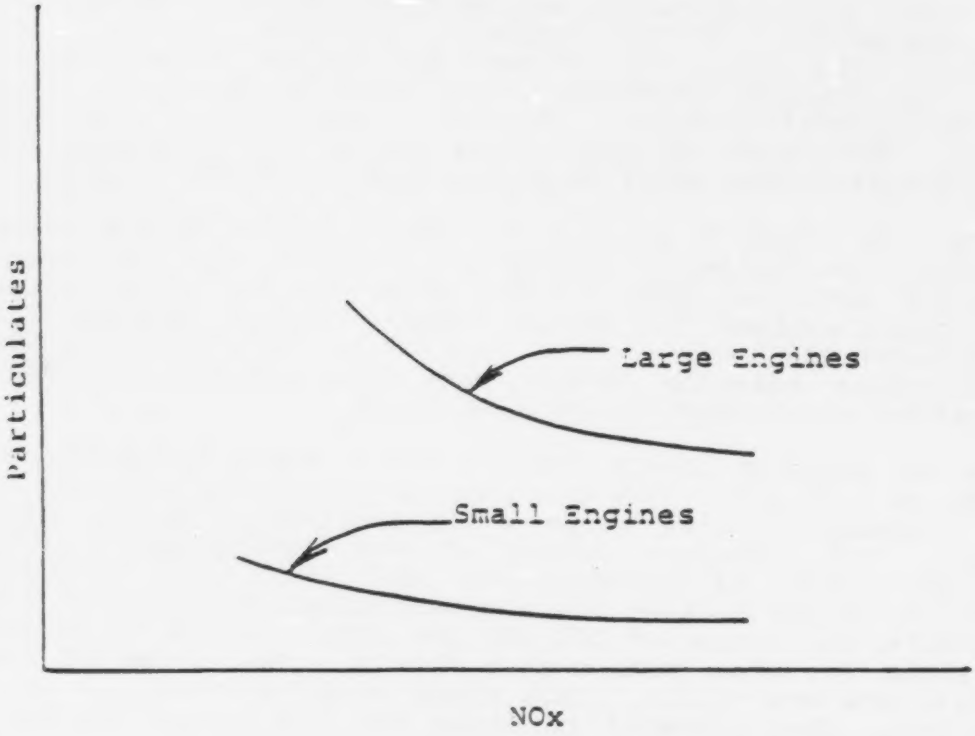
First, in order to account for the shifting of the curve that occurs with changes in engine displacement, the 1983 model year engine families were divided into the following three groups: small engines (1.6 to 1.8 liters), medium engines (2.0 to 2.8 liters) and large engines (3.0 to 5.7 liters). The NOx and particulate emission levels were then plotted for each configuration within each engine size group.

The NOx emission levels for the small engine group ranged from 0.80 to 1.29 g/mi. The distribution of points appeared to have a slightly negative slope which regression of the data confirmed. The emission levels of one configuration (VW, engine family AZ8, A3 transmission, 2500 lbs.) were excluded from the regression because the NOx/particulate combination was well outside the range of all of the other values including other values for that same engine family. The slope of the regression line was -0.033. This slope is quite small, as will be seen later when compared to those for the larger engines, and is generally in line with what would be expected for small engines.[1] Therefore, it was used to predict changes in particulate emission levels resulting from changes in NOx emission levels below an absolute NOx level of 1.35 g/mi. It was assumed that no further reduction in particulate would occur for NOx emission levels greater than 1.35 g/mi (i.e., the slope was considered to be zero).[1]

A NOx emission level of 1.35 g/mi was chosen as the reference point to change slopes for the small engine group (and the other two groups) for two reasons. First, the great majority of current LDDVs have NOx emissions less than 1.35 g/mi. Since the slopes obtained by the regression of the data

Figure 1-1

Generalized Shape of NOx/Particulate Tradeoff
Curves Illustrating the Shifting Effect of Engine Displacment



are most appropriate within the distribution of data, it was decided to limit the applicability of the regressions to this level. Second, 1.35 g/mi is approximately the engineering objective (or low mileage target (LMT)) for the NOx standard of the base scenario (1.5 g/mi). (The NOx standard of 1.5 g/mi minus a 10 percent safety margin and divided by a deterioration factor (DF) of 1.000 (which is typical for diesel NOx emissions) yields a LMT of 1.35 g/mi.)

The plot of the emission levels for the medium engine group, whose NOx values ranged from 0.87 to 1.37 g/mi, appeared to have a greater negative slope than the small engine group. Regression of the data confirmed this, showing a slope of -0.201. As this slope appeared reasonable for engines of this size, based on the limited information available on current NOx/particulate tradeoff curves[1] and the fact it was larger than the slope for the small engines, this slope was used to predict the change in particulate emission levels for changes in NOx emission levels below 1.35 g/mi NOx. Again, two configurations' emission levels (M-B, 2.4L, both transmissions) were not used in the regression because they definitely appeared to be outliers. The slope of the tradeoff curve for NOx emission levels greater than 1.35 g/mi was somewhat arbitrarily reduced by one-half to -0.100, since it is known that the curve becomes flatter at higher NOx levels, but by an unknown magnitude.

For the large engines, the slope of the tradeoff curve for NOx values below 1.35 g/mi was not based on a regression of the data, but was simply estimated to be -0.400 based on known tradeoff curves for large, albeit older, engines.[1] This was necessary because there were only 8 data points for the large engines and no correlation existed. The slope for NOx values greater than 1.35 g/mi was also estimated using engineering judgment and was set at -0.100. At first a slope of -0.200 was estimated, based on the judgment that this slope should be steeper than that for the medium-size engines. However, this produced some unrealistically low particulate values at the higher NOx values being examined in the NOx sensitivity analysis, so -0.100 was chosen instead.

C. Engine Configuration's Low-Mileage Targets and Standard Levels

Using the slopes of the tradeoff curves determined above and the data in Table 1-1, the particulate low-mileage target (LMT) at 1.35 g/mi NOx was calculated for each configuration. The particulate standard level for each configuration was then calculated for these particulate LMTs. This was done by multiplying the particulate LMT by the appropriate 50,000 mile deterioration factor (DF) and the appropriate safety margin. Both factors are explained below.

The particulate DF used for each configuration was the certification DF for the 1983 model year except in three instances. The three exceptions were engine families* with DFs much greater than the other 18 engine families. Fifteen of the 21 total engine families had particulate DFs less then 1.10. Another three engine families had particulate DFs between 1.10 and 1.15. The last three engine families had particulate DFs greater than 1.24. It was concluded that the manufacturers of these last three engine families could lower the DFs to at least the 1.15 level if a more stringent particulate standard required them to do so. Therefore, for the purposes of this study, a DF of 1.15 was assumed for each of those three engine families.

The safety margins necessary for calculating the particulate standard levels from each particulate LMT were determined using the methodology developed for past EPA rulemakings.[2] That methodology requires a coefficient of variation (COV) for production-line vehicles and the number of prototype vehicles tested before a manufacturer fixes its design. Results from EPA's Selective Enforcement Audit (SEA) testing program[3] indicate that the LDDV particulate COV is slightly less than 0.13. Also, the number of prototype vehicles to be built and tested was presumed to equal the maximum considered in the methodology (seven), since the engine technology exists today and manufacturers will have more than sufficient data upon which to base their LMTs. Thus, the safety margin as interpolated from the table[2] would be seven percent. However, since available SEA test data on LDDVs is limited and the particulate COV may increase somewhat with more stringent NOx and/or particulate standards, a somewhat larger safety margin of 10 percent was used for this study.

The particulate standards achievable by each configuration are shown in Table 1-2. An industry-wide, non-averaging, non-trap, non-technology forcing particulate standard can be determined by simply identifying the configuration with the highest particulate standard listed in Table 1-2. Thus, for the NOx standard of 1.5 g/mi, such a particulate standard would be 0.43 g/mi (M-B, 3.0L engine).

It should be noted that this highest emitting configuration, as well as the next three highest emitting configurations, seem to be technology outliers. Three out of four of these configurations are Mercedes-Benz (M-B) vehicles. When the emissions of these M-B vehicles are compared to those of other similarly sized vehicles, one finds that the M-B

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* Engine families, rather than configurations, are considered here because DFs are only determined on an engine family basis and are applied to all configurations within that engine family.

Table 1-2

Achievable Non-Trap, LDDV Particulate Standards Under the 1.5 g/mi NOx Standard

Manufacturer	Engine Family	Transmission	Inertia Weight Class (lb)	Displacement (liters)	Part. Std. Assuming a 1.5 g/mi NOx Std. (g/mi)
General Motors	Z90	M5	2,500	1.8	.20
	Z90	L3	2,500	1.8	.15
	ZK7	L3	3,000	4.3	.12
	ZK7	L3	3,500	4.3	.17
	ZT8	L3	3,500	4.3	.19
	ZT7	L3	4,000	5.7	.33
	ZT7	L3	4,500	5.7	.36
	ZT7	L4	4,000	5.7	.35
	ZT7	L4	4,500	5.7	.42
Volkswagen	AAO	M4	2,250	1.6	.17
	AAO	M5	2,250	1.6	.21
	AAO	A3	2,250	1.6	.20
	JAO	M5	2,500	1.6	.21
	JAO	A3	2,500	1.6	.19
	AZ8	M5	2,250	1.6	.27
	AZ8	M5	2,500	1.6	.24
	AZ8	A3	2,500	1.6	.26
	RA5	S4	2,250	1.6	.20
	BZX	A3	2,750	1.6	.20
	BZX	M5	2,750	1.6	.24

Table 1-2 (cont'd)

Achievable Non-Trap, LDDV Particulate Standards Under the 1.5 g/mi NOx Standard

Manufacturer	Engine Family	Transmission	Inertia Weight Class (lb)	Displacement (liters)	Part. Std. Assuming a 1.5 g/mi NOx Std. (g/mi)
Nissan	AF8	M4	2,250	1.7	.20
	AF8	M5	2,250	1.7	.24
	AF8	M5	2,500	1.7	.28
	AF8	A3	2,250	1.7	.29
	AF8	A3	2,500	1.7	.27
	AFO	M5	3,500	2.8	.22
	AFO	L4	3,500	2.8	.28
Mercedes-Benz	501	M4	3,500	2.4	.41
	501	A4	3,500	2.4	.37
	508	A4	4,000	3.0	.43
Isuzu	CD7	M4	2,500	1.8	.22
	CD7	M5	2,500	1.8	.20
	CD7	M5	2,750	1.8	.21
	CD7	A3	2,750	1.8	.19
Audi	BZ7	M5	2,750	1.6	.24
	BZ7	A3	2,750	1.6	.19
	CZ3	A3	3,000	2.0	.19
	BAX	M5	2,500	1.6	.25
Peugeot	AA1	M5	3,500	2.3	.24
	AA1	A3	3,500	2.3	.26
	BA3	M4	3,500	2.3	.25
	BA3	A3	3,500	2.3	.36
Volvo	AY2	M4	3,500	2.4	.36
	AY2	A3	3,500	2.4	.32
	TB0	M5	3,500	2.4	.32
	TB0	A3	3,500	2.4	.25

vehicles emit significantly more particulate. Thus, it appears that M-B has not yet implemented the kinds of combustion chamber and injection modifications that have been made by others (e.g., General Motors). Presumedly, M-B could do this if it became necessary. The fourth configuration is a General Motors vehicle powered by their 5.7 L engine (L4 transmission, 4500 lbs.). It has been rumored that this engine will be eliminated sometime in the next few years, primarily due to market considerations, as well as to the fact that this is their highest emitting engine for both NOx and particulate. If these four configurations were excluded from consideration here, the non-averaging, non-technology forcing particulate standard could be 0.36 g/mi, 15 percent lower than the 0.43 g/mi level mentioned above.

D. Non-Trap "Averaging" Standards

The previous discussion presented the methodology used to estimate particulate LMTs and standard levels for each configuration under a NOx standard of 1.5 g/mi. From those results an industry-wide, non-averaging, non-trap, non-technology forcing particulate standard could be selected. That standard was based on the assumption that every LDDV would need to be at or below the standard (i.e., non-averaging). In this situation, most vehicles could increase their particulate emissions up to the level of the worst-case vehicle and still be in compliance. While we would not expect such a situation to occur to this extreme, it is possible that NOx control and fuel economy incentives could lead to increased particulate emissions if the particulate standard allowed it.

One way to significantly increase the probability that industry-wide particulate emissions would not increase beyond present levels and yet still set a non-technology forcing standard would be to implement a corporate average standard (see the introduction to the study for an explanation of emissions averaging). Under this approach the non-trap, non-technology forcing standard would be numerically lower than that determined in the previous discussion since each manufacturer could "average" its high emitters with its low emitters. Because of this, it becomes more difficult for a manufacturer to increase the emissions of its low emitters since these emissions are factored into its corporate average emission level and are no longer irrelevent. Thus, while averaging has been considered in the past only for trap-based particulate standards, it also has a benefit for non-trap standards.

Table 1-3 shows the currently achievable non-trap, non-technology forcing particulate standard for each manufacturer under averaging. These standards were calculated by sales weighting the achievable particulate standards for each manufacturer's LDDV configurations listed in Table 1-2. Sales for each configuration were obtained from the

Table 1-3

Achievable Non-Trap Particulate Standards (under "averaging")

Manufacturer	Assuming a 1.5 g/mi NOx Standard (g/mi)
General Motors	.29*
Volkswagen	.20
Nissan	.26
Mercedes-Benz	.42
Isuzu	.20
Audi	.20
Peugeot	.26
Volvo	.29

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* This level becomes 0.16 g/mi if GM's 5.7-liter engine is discontinnued and its sales are replaced by their 4.3-liter engine.

manufacturers' 1983 estimated Federal sales required by the fuel economy program known as Corporate Average Fuel Economy (CAFE).* If the worst-case manufacturer's (i.e., Mercedes-Benz) particulate averaging standard level became the averaging standard for the industry, then as Table 1-3 shows, the particulate averaging standard would be 0.42 g/mi. This level is not significantly lower than the 0.43 g/mi non-averaging standard of the previous section due to what appears to be excessively high emission levels of the worst-case manufacturer's engines. If this worst-case manufacturer is treated as a technology outlier, then the corporate average for General Motors and Volvo would set the industry-wide, particulate averaging standard at 0.29 g/mi. This level is 33 percent lower than the non-averaging standard of the previous section (0.43 g/mi). A non-trap, non-technology forcing, particulate averaging standard of 0.29 g/mi would moderate the risk that manufacturers of small LDDVs, which are low particulate emitters, might substantially increase particulate emissions from these vehicles.

It is interesting to note what would happen to GM's corporate average particulate level if it discontinued production of its 5.7-liter engine. This could happen if the long-range trend towards increased fuel economy eliminated the "big" cars of today whereupon the need for the 5.7-liter engine would also be eliminated. Assuming that the vehicles which would have had the 5.7-liter engine received instead GM's 4.3-liter engine, GM's average particulate standard level would drop from 0.29 g/mi as shown in Table 1-3 to 0.16 g/mi. Furthermore, since GM's estimated sales comprise about 60 percent of the total LDDV estimated sales, lowering GM's average particulate standard level by this 45 percent would lower total LDDV particulate emissions substantially. However, the non-trap, non-technology forcing, averaging particulate standard based on the second highest emitter would remain at 0.29 g/mi (Volvo).

E. Determination of the Percent of Trap-Equipped Vehicles

Thus far, this analysis has been concerned with only those particulate standards achievable without the use of trap-oxidizer systems. It will now consider the use of traps as a particulate control strategy. Here the focus of the analysis will differ from that of the previous section. Instead of determining achievable particulate standards under various scenarios which assume some percent usage of traps (in the previous case, zero), this discussion will assume a particulate standard of 0.20 g/mi (the base scenario) and then determine the percentage of the LDDV fleet requiring traps in order to achieve this standard. Emissions averaging will be assumed to apply as the Agency expects to soon finalize a particulate averaging program in conjunction with the 0.2 g/mi standard (proposed in 46 FR 62608).

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* These projections are confidential and are not presented here. The presentation of the resultant corporate emission average does not divulge the pertinent information contained in the projections (i.e., absolute sales).

Two types of traps were considered for compliance with the 0.20 g/mi particulate standard. One is the wire mesh type and the other is the ceramic type. EPA's report[4] on the feasibility of trap oxidizers indicated that both types appear to have good durability characteristics. The ceramic type of trap was tested by Southwest Research Institute for EPA[5] and the wire mesh type was tested at this same facility for Johnson-Matthey, Inc.[6] These testing programs indicated that the deterioration for both types of traps was negligible and, therefore, a DF of 1.00 was used here. The EPA[4] report discusses each of the two traps in detail and concludes that the efficiency of the ceramic trap is about 70-90 percent while the efficiency of the wire mesh trap is about 50-80 percent. As the durability test of the ceramic trap referred to above showed an 85 percent efficiency, that figure will be used here. For the wire mesh trap, 65 percent will be used as a reasonable mean efficiency for a typical trap. This analysis will use these percent efficiencies to determine the tail pipe emission levels from engine-out emission levels. (For simplicity, mixing use of both trap types was avoided.)

The methodology used to calculate the percentage of trap-equipped vehicles for each type of trap and manufacturer is straightforward. First, each configuration's estimated sales was multiplied by that configuration's non-trap standard level taken from Table 1-2. These results were then summed to obtain the total number of vehicle-grams per mile (veh-g/mi) from which each manufacturer would begin its control efforts. Next, each manufacturer's total estimated sales were multiplied by 0.20 g/mi to give the total veh-g/mi that the manufacturer would be allowed under each particulate averaging standard. The difference between the two figures is the amount of control each manufacturer needs to achieve. It was assumed that a manufacturer would put traps on its highest emitters first because the g/mi reduction achieved is highest for those vehicles.

To determine the number of veh-g/mi saved by putting a trap on a given configuration, that configuration's particulate LMT was first multiplied by one minus the trap efficiency and then multiplied by that configuration's particulate DF. This result was then transformed into a new particulate standard level by adding a safety margin of 10 percent or 0.02 g/mi, whichever was greatest, because 0.02 g/mi was considered the minimum acceptable safety margin. The new particulate standard level was then multiplied by the estimated sales for that configuration to give the new total veh-g/mi emitted. The difference between the total veh-g/mi without traps and the total veh-g/mi with traps was counted as controlled veh-g/mi. Calculations were made for each configuration until the controlled veh-g/mi equalled or exceeded the amount of veh-g/mi that the manufacturer needed to control in order to meet the 0.2 g/mi particulate standard. For the most part, only enough traps were assumed installed to just meet the standard. However, in a few instances where the percentage of traps on a given configuration approached 80 percent it was assumed that the whole configuration would be equipped with traps.

The results of these calculations are shown in Table 1-4. Assuming ceramic traps, 22 percent of all LDDVs would require traps to comply with the 0.20 g/mi standard. Assuming wire mesh traps, this figure increases to 30 percent.

III. Light-Duty Diesel Trucks

The methodology used to estimate the non-trap, non-technology forcing, particulate standards for each LDDT configuration was the same as that used for LDDVs. The current particulate emission test levels for small LDDTs (i.e., engine displacements from 1.6 to 2.3 liters), which were obtained from certification test results, were used to calculate the particulate standard levels shown in Table 1-5. The NOx emission levels of the majority of these configurations were between 1.35 and 1.7 g/mi. Since the NOx/particulate tradeoff curve for small LDDV engines was flat in this region, no adjustment was made to the small LDDT certification values in deriving the particulate LMTs. For the full-size LDDTs (i.e., engine displacements of 6.2 liters) the current particulate emission test levels were adjusted to their equivalent at 2.05 g/mi NOx using the same NOx/particulate tradeoff curve (slope of -0.100) as that used for large LDDVs. (A LDDT NOx level of 2.05 g/mi under a 2.3 g/mi NOx standard is equivalent to the 1.35 g/mi NOx level for LDDVs. Also, the -0.1 slope curve was the only one needed since the certification NOx emission levels of the full-size LDDTs were all above 1.5 g/mi.) As shown in Table 1-5, the industry-wide, non-trap, non-technology forcing particulate standard without averaging would be 0.40 g/mi.

Table 1-6 presents each LDDT manufacturer's non-trap, non-technology forcing, particulate standard under the averaging concept. These levels were calculated using the same methodology as was previously described for LDDVs. The highest average particulate level is for Mitsubishi at 0.39 g/mi. Note that this level is well above that for GM (0.28 g/mi), which only produces full-size LDDTs. Thus, if Mitsubishi were considered controlling, there is very little difference between the non-averaging and the averaging non-trap standard for LDDTs.

Table 1-4

Percentage of LDDV Sales Requiring Traps Under Various Particulate Standards (assumes "averaging")

1.5 g/mi NOx Standard

Manufacturer	0.20 g/mi Particulate Standard with Ceramic Trap	0.20 g/mi Particulate Standard with Wire Mesh Trap
General Motors	26.8	36.2
Volkswagen	0	0
Nissan	25.6	33.4
Mercedes-Benz	55.5	79.6
Isuzu	0	0
Audi	2.2	2.9
Peugeot	30.3	39.5
Volvo	34.4	44.2
Industry-Wide Sales-Weighted Percentage	22.3	30.2

Table 1-5

Achievable Non-Trap, LDDT Particulate Standards Under the 2.3 g/mi NOx Standard

Manufacturer	Engine Family	Transmission	Inertia Weight Class (lbs.)	Displacement (liters)	Part. Std. Assuming a 2.3 g/mi NOx Std. (g/mi)
Small LDDTs:					
Ford	AG5	M4	3,000	2.2	.29
Isuzu	CD3	M4	2,750	2.2	.28
		M4	3,000	2.2	.26
		M5	3,000	2.2	.25
Nissan	AF9	M5	3,000	2.2	.35
Mitsubishi	FD0	M5	3,000	2.3	.39
		M5	3,500	2.3	.38
Toyota	BB8	M5	3,000	2.2	.17
	FF9	M5	3,000	2.2	.25
Volkswagen	PA2	M4	2,250	1.6	.26
		M5	2,250	1.6	.38
	VA9	M5	3,500	1.6	.27
		M5	4,000	1.6	.33
Toyo Kogyo	KK9	M5	3,000	2.2	.29
Full-Size LDDTs:					
General Motors	Z40	M4	4,500	6.2	.32
		L4	4,500	6.2	.35
		M4	5,000	6.2	.26
		L4	5,000	6.2	.28
		M4	5,500	6.2	.40
		L4	5,500	6.2	.26
		L4	6,000	6.2	.36

Table 1-6

Achievable Non-Trap Particulate Standards Under "Averaging"

Manufacturer	Assuming a 2.3 g/mi NOx Standard
Small LDDTs:	
Ford	.29
Isuzu	.25
Nissan	.35
Mitsubishi	.39
Toyota	.19
Volkswagen	.31
Toyo Kogyo	.29
Full-size LDDTs:	
General Motors	.28

As in the LDDV case, the percentage of LDDTs requiring trap-oxidizer systems under the base scenario (0.26 g/mi particulate standard) was determined. The methodology used to determine this percentage was the same as for LDDVs except that small and full-size LDDTs were considered separately. This was done because the ratio of sales of small to full-size LDDT sales is expected to change significantly by the mid-to-late 1980s. A study[7] by Jack Faucett Associates (JFA) projects that in 1987, 86.5 percent of all new LDDT sales will be full-size while only 13.5 percent will be small. Manufacturers' LDDT sales estimates for the 1983 model year indicate that currently full-size LDDTs represent about 55 percent of all LDDT sales. Thus, a substantial change is expected to occur over the next several years. Therefore, the percent of traps required by each LDDT-size group was weighted according to the findings by JFA and then combined into a single LDDT percentage.

Table 1-7 presents the percentage of sales for each manufacturer that would require ceramic traps under the 0.26 g/mi particulate standard. For simplicity these calculations were not done for the wire mesh trap, as the effect of using wire mesh traps instead of ceramic traps was estimated in Section II.E. for LDDVs and, given present data, the ceramic trap appears to have advantages over the the wire mesh trap in terms of cost and trapping efficiency. If the percentages of wire mesh traps required per manufacturer were desired, they could be easily approximated by applying the ratio of the percent of LDDVs which would require wire mesh traps to the percent of LDDVs which would require ceramic traps (see Section II.E.)

From Table 1-7, the industry-wide percentage of sales that would require ceramic traps under the base scenario is estimated to be 7.6 percent.

IV. Heavy-Duty Diesels

A. Current Emission Level and Non-Trap Standards

Currently there is no particulate standard for heavy-duty diesel engines (HDDEs). Therefore, there are no certification test data from which to determine the current levels of HDD particulate emissions. However, there has been a substantial amount of HDD particulate testing over EPA's new transient cycle by both EPA and the industry. Table 1-8 contains particulate and NOx emission data from manufacturers' production and development tests,[8] the EMA/EPA HDD "round-robin" testing program,[9] and EPA's original diesel transient baseline[10] (for those engines for which more recent data are not available). Although data are not available for every HDD engine family, a large majority of sales is represented. Sales weighting the data in Table 1-8 indicated

Table 1-7

Percentage of LDDT Sales Requiring Traps Under Various Particulate Standards (assumes "averaging")

	0.26 g/mi Part. Std. with Ceramic Trap
Small LDDTs:	
Ford	12.3
Isuzu	0.0
Nissan	31.9
Mitsubishi	40.1
Toyota	0.0
Volkswagen	15.4
Toyo Kogyo	11.5
Full-Size LDTs:	
General Motors	6.9
Industry-Wide Sales-Weighted Percentage	7.6

Table 1-8

Low-Mileage, Transient Emissions From Current Heavy-Duty Diesel Engines

Manufacturer/ Engine	Particulate (g/BHP-hr)	NOx (g/BHP-hr)
Caterpillar		
3208 DINA	0.65	7.8
3208 DIT	0.59	10.0
3406 DITA	0.52-0.71	7.9-8.4
3406 PCTA	0.37	5.4
3306 DITA	0.73	9.0
3306 PCTA	0.50	4.8
Cummins		
NTC 290	0.59	8.3
NTC 350	0.58-0.70	7.2-9.0
NTC 350 (Big Cam)	0.40	6.8
NTCC 240	0.77	4.8
NTCC 400	0.85	5.3
NH 250	0.52-0.83	6.8-6.9
VTB-903	0.67	5.2
Daimler-Benz		
OM 344A	0.81	5.1
OM 362A	0.45	6.7
Detroit Diesel		
8V-71N	0.79	5.7
8V-71TA	0.35-0.43	6.7-7.6
6V-92TA	0.55-0.67	5.8
8V-92TA	0.46	7.8
8.2-T	0.43	5.0-5.9
International Harvester		
DT-466B	0.53	5.7
DTI-466B	0.67	4.2
	0.31-0.36	5.6-5.7
Mack		
ETAZ-676	0.58	5.2
ETSX-676	0.63-0.69	5.2
ETSZ-676	0.59	6.9

an average particulate emission level of around 0.60-0.65 g/BHP-hr. After allowing for some deterioration (these engines were almost entirely new), it is estimated that today's HDDs emit at an average of 0.7 g/BHP-hr in-use.

While the 0.7 g/BHP-hr level is appropriate for today's engines, future HDDs should be able to reach somewhat lower particulate levels with relatively minor engine modifications and recalibrations. The impetus to control HDD particulate (other than the particles constituting "smoke" at certain extreme engine operation modes) has not yet occurred since there has been no particulate standard. With a standard, however, some reduction in particulate emissions should occur. For example, in its comments to the HDD particulate NPRM,[11] Caterpillar recommended a future non-trap standard of 0.6 g/BHP-hr, including DF and safety margin. For the purposes of this analysis, this level will be used as the non-trap, non-technology forcing, HDDE particulate standard to be implemented sometime in the 1987-88 timeframe. Also, for the purposes of this analysis, we have assumed that this standard would be implemented in 1988. Thus, without the use of trap-oxidizers, HDDEs will be projected to emit at 0.7 g/BHP-hr through 1987 and at 0.6 g/BHP-hr thereafter.

B. Standard Level With Traps

The trap-based HDDE particulate standard of the base scenario is 0.25 g/BHP-hr. This level was proposed by the Agency in its HDDE particulate NPRM (46 FR 1910). The percentage of HDDEs that would require traps under this standard is 100 percent because it was proposed as a non-averaging standard and all HDDEs currently emit substantially above 0.25 g/BHP-hr. (The effect of averaging will be considered later in this section.)

The 0.25 g/BHP-hr standard requires a 60 percent reduction in particulate emissions from the 0.6 g/BHP-hr non-trap level mentioned above. Both the ceramic trap and the wire mesh trap have efficiencies greater than 60 percent. Under the base scenario without averaging, it has been assumed that manufacturers would only apply traps of the required efficiency regardless of the type of trap used. This is to say that even if ceramic traps were applied, there would be sufficient impetus to reduce efficiency below that achievable (e.g., to increase regeneration intervals and reduce backpressure and fuel economy penalties) if the standard were more stringent, that only the efficiency actually necessary, with a reasonable safety margin, would be applied. This efficiency has been assumed to be 65 percent. Applying this 65 percent efficiency to the engine-out emission standard level of 0.6 g/BHP-hr, results in tailpipe emissions of 0.21 g/BHP-hr under the base scenario. This is somewhat lower than the required 0.25 g/BHP-hr, but is appropriate because it is believed that HDD manufacturers will desire a somewhat larger safety margin due to the variety of HDD application and the absence of averaging.

If averaging were implemented along with the 0.25 g/BHP-hr standard for HDDEs, the percentage of vehicles requiring traps would drop from 100 to about 70 percent. In this case, we have assumed that manufacturers would utilize the full 85 percent efficiency of the ceramic trap in order to take full advantage of averaging.

References

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2. "Regulatory Analysis and Environmental Impact of Final Emission Regulations for 1984 and Later Model Year Heavy-Duty Engines," U.S. EPA, OANR, OMS, ECTD, SDSB, pp. 184-86, December 1979.

3. This Data is Publicly Available From the U.S. EPA Selective Enforcement Section, Manufacturer's Operations Division, Office of Mobile Sources.

4. "Trap-Oxidizer Feasibility Study," U.S. EPA, OANR, OMS, ECTD, SDSB, March 1982.

5. "Light-Duty Diesel Organic Material Control Technology Investigation Program," EPA Contract No. 68-03-2873, Monthly Progress Report No. 34, August 10, 1982.

6. Letter from B. E. Enga, Johnson-Matthey, Inc., to Anne M. Gorsuch, Administrator, U.S. EPA, Regarding the 1985 Light-Duty Diesel Particulate Standards, January 25, 1982 (EPA Docket A-81-20, II-D-75).

7. "The Impact of Light-Duty Diesel Particulate Standards on the Level of Diesel Penetration in the Light-Duty Vehicle and Light-Duty Truck Markets," Jack Faucett Associates, For U.S. EPA, Contract No. 68-01-6375.

8. This Data Was Submitted by Heavy-Duty Diesel Engine Manufacturers As Comments to EPA's Heavy-Duty Diesel Particulate NPRM (46 FR 1910) and Can Be Found In EPA Public Docket No. A-80-18.

9. "EMA/EPA Heavy-Duty Diesel Engine Cooperative Test Program," EPA Public Docket No. A-80-18, November 1982.

10. "Emissions From Heavy-Duty Engines Using The 1984 Transient Test Procedure, Volume 2 - Diesel," U.S. EPA, OANR, OMS, EPA-460/3-81-031, July 1981.

11. This Data is Contained In Caterpillar Tractor Company's Comments to the Heavy-Duty Diesel Particulate NPRM (46 FR 1910). Caterpillar's Comments Can Be Found In EPA Public Docket No. A-80-18.

CHAPTER 3

AIR QUALITY IMPACT AND POPULATION EXPOSURE

I. Introduction

In an attempt to place the impact of the urban emission estimates of the previous chapter in a better perspective for assessing both health and welfare impacts, this chapter estimates the air quality impact of and population exposure to diesel particulate emissions in 1995 under the various diesel sales and control scenarios outlined in Chapter 1. This is accomplished in four sections.

The first section outlines and uses a methodology for deriving nationwide average diesel particulate emission factors for urban areas in 1995. These scenario-specific nationwide average diesel particulate emission factors become the primary input to the following three sections.

The second section of this chapter uses atmospheric lead monitoring data as a surrogate to estimate atmospheric levels of diesel particulate in 1995 under the various scenarios. This analysis will provide estimates of ambient diesel particulate concentrations at one or two particular monitor locations in a large number of U.S. cities, with basic input to the model consisting of national fleet-wide averages. These 1995 particulate concentrations are then compared both to each other and to 1980 levels.

The third section is concerned with a similar analysis of four types of localized areas which are particularly sensitive to motor vehicle emissions. These microscale areas include urban expressways, street canyons and enclosed spaces such as parking garages and roadway tunnels.

While yielding estimates of diesel particulate concentrations in particular locations, neither the urban nor localized air quality analyses address overall population exposure as people move from location to location within an urban area. This is done in the fourth section of this chapter by estimating the actual exposure of individuals to these concentrations; these results can then be used to assess the cancer risk associated with diesel particulate. The exposure analysis uses a CO exposure model which was developed by EPA's Office of Air Quality Planning and Standards (OAQPS) for use in evaluating alternative CO National Ambient Air Quality Standards (NAAQS). Since studies show that the great majority of CO concentrations in the atmosphere are mobile source related, it is felt that CO is reasonably representative of vehicle pollutant trends and, therefore, can serve as an acceptable surrogate for diesel particulate matter in exposure modeling. It should be noted that those sources of CO which are not motor vehicle related, such as indoor sources, are removed from the model for this analysis.

A final section is included in support of the theories behind the air quality analysis to be performed in this chapter. By examining trends in historical emissions versus ambient concentrations over a period of time, a direct correlation is demonstrated for both lead and CO; this supports the rollback theory that emissions can be used to predict ambient concentrations. Also included in the final section is a comparison of this chapter's air quality projections to those contained in a recent EPA study. Ambient diesel particulate concentrations for 1980 are calculated using the models contained in both reports; upon comparison of results, their similarity demonstrates support for the lead surrogate model used here.

It should be remembered that the methodologies described in Section II through IV utilize the ambient measurement of other pollutants (lead and CO) to estimate the future year (1995) concentrations of diesel particulate. None of the models are based on actual measurements of urban levels of diesel particulate. As with any indirect analysis method, the absolute accuracy of this methodology is not well known because direct measurements of the pollutant of interest in urban areas cannot be made. (It is difficult to distinguish diesel particulate from other airborne carbonaceous particulate.) For this very reason, these surrogate techniques are probably the most suitable approaches currently available for projecting diesel particulate concentrations in numerous areas around the U.S.

It is also important to point out that national-average input data are used throughout this analysis. Such parameters as the lead content of gasoline, vehicle mix, diesel market penetration, VMT growth, pollutant dispersion characteristics, temperature variations, etc., can vary both regionally and locally; changes in these parameters would, in turn, have an effect on the projected diesel particulate concentrations. Therefore, care should be taken not to overemphasize the individual diesel particulate concentrations projected for specific cities (listed later in the chapter). More accurate estimates for particular cities could be made if the specific traffic characteristics for that city were used; however, an individual city-by-city analysis was beyond the scope of this report. Instead, the primary purpose of the modeling efforts in this chapter is to estimate the effect various levels of emission control will have on future urban concentrations of diesel particulate across the nation as a whole.

II. Nationwide Diesel Particulate Emission Factors

The first step in estimating either annual average ambient particulate levels in U.S. cities or the nationwide average urban population exposures is to derive fleet-wide urban diesel particulate emission factors for urban areas for each of the four scenarios. To do this, the procedures outlined in Chapter 2 are repeated to determine the average diesel particulate emission factor for each vehicle class and scenario as shown in Table 3-1 (reproduced from Table 2-8 of Chapter 2). The emission factors for each vehicle category in a particular scenario are then combined according to the weighting of their 1995 urban VMT, which can be derived from the projected VMT data in Table 3-1.

Table 3-2 again shows the particulate emission factors for each vehicle category/scenario, the derived urban VMT breakdown for 1980 and 1995, and the fleet-wide, urban particulate emission factors for each scenario. Also shown is a breakdown of each vehicle class' contribution to urban emissions under each scenario.

III. Urban Air Quality Analysis

Since diesel particulate is not easily distinguishable from other carbonaceous particulate, air quality monitoring data are not presently available for diesel particulate, especially under the conditions expected to exist in 1995. Thus, any method for estimating diesel particulate air quality impacts must use some measurable surrogate in the ambient air that is directly relatable to automobile emissions. Various studies in the past have used such substances as lead or CO to provide a link between vehicle emissions and air quality. Once this link is established, then vehicle emissions of the surrogate substance are related to diesel particulate emissions, resulting in an estimate of diesel particulate air quality impacts.

The fairly strong correlation between ambient concentrations of lead and carbon monoxide documented in various published reports [1-4] supports the theory that both are representative of mobile source contributions to air quality. Observed concentrations of Pb and CO in Los Angeles in 1980 [2] show the two pollutants to exhibit very similar monthly and seasonal variations; a linear regression of matched concentration pairs of the two pollutants yields an r^2 value of 0.7980. Although the actual ratio of CO/Pb varies from study to study because of yearly changes in gasoline lead contents, the coefficient of variation from the mean ratio is fairly consistent, ranging from 20 to 26 percent [1,2,4]. It

Table 3-1

Weighted Emission Factors and Projected VMT

	LDV	LDT	MDV/LHDV	HHDV
Weighted Emission Factor (g/mi)				
Calendar Year 1980:				
All Scenarios	0.0059	0.0074	0.0676	1.913
Calendar Year 1995:				
Best Estimate Diesel Sales				
Relaxed Scenario	0.0272	0.0760	0.4130	1.6589
Base Scenario	0.0205	0.0711	0.2020	0.8499
Worst-Case Diesel Sales				
Relaxed Scenario	0.0606	0.1170	0.8088	1.7025
Base Scenario	0.0460	0.1092	0.3864	0.8753
Projected Nationwide VMT (10^9 miles):[18]				
1980	1,118	208.9	38.8	73.8
1986	1,220	241.0	41.2	85.7
1995	1,537	329.5	55.2	117.9
Urban Fraction of VMT: (all years)	59.4	48.8	48.8	26.9
Urban VMT (10^9 miles):				
1980	664.1	101.9	18.9	19.9
1986	724.7	117.6	20.1	23.1
1995	913.0	160.8	26.9	31.7

Table 3-2

Derivation of National Average
Diesel Particulate Emission Factors (g/mi)

	Urban VMT Breakdown (%)		Emission Factor (g/mi)				
				Best Estimate Sales		Worst Case Sales	
	1980	1995	1980	Relaxed	Base	Relaxed	Base
LDV	82.5	80.6	0.0059	0.0272	0.0205	0.0606	0.2460
LDT	12.7	14.2	0.0074	0.0760	0.0711	0.1170	0.1092
MDV/LHDV	2.3	2.4	0.0676	0.4132	0.2022	0.8088	0.3864
HHDV	2.5	2.8	1.9130	1.6589	0.8499	1.7025	0.8753
Fleet-Average Urban-VMT Weighted Emission Factor			0.0552	0.0891	0.0554	0.1324	0.0863
Vehicle Class Contribution to Urban Emissions %:							
LDV			8.8%	24.6%	29.9%	36.9%	43.0%
LDT			1.7%	12.1%	18.2%	12.5%	18.0%
MDV/LHDV			2.8%	11.0%	8.7%	14.5%	10.7%
HHDV			86.7%	52.3%	43.2%	36.0%	28.4%

should be noted that the L.A. study [2] showed a slightly greater correlation between elemental carbon (EC) and both Pb and CO; r^2 values for EC/Pb and EC/CO were 0.8907 and 0.8881, respectively. The above correlations, along with the knowledge that approximately 90 percent of all fine elemental carbon (less than 10 um diameter) in the atmosphere is estimated to be due to mobile sources [5], leads to the conclusion that elemental carbon could also possibly serve as a surrogate for diesel particulate in projecting future ambient air quality. To date, however, the most commonly used surrogates are lead and carbon monoxide.

One methodology, which has been used in the past by GM and EPA, uses lead as a surrogate for diesel particulate.[6,7] This type of analysis uses historical data from urban sites in the national urban lead monitoring network as an index of mobile source pollutant levels. An estimate is made of the fleet's automotive lead emission factor which caused the observed ambient lead levels, and is compared to the expected diesel particulate emission factor. Very generally speaking, if diesel particulate emissions in 1995 are expected to be twice automobile lead emissions in 1975, for example, then ambient diesel particulate concentrations in 1995 can be expected to be twice the 1975 ambient lead concentrations. In this case, 1975 monitoring data was chosen over more recent data to avoid, for the most part, the errors associated with estimating the leaded/unleaded vehicle mix.

The basic mathematical expression of this methodology is:

$$C(D)_{1995} = \frac{E(D)_{1995}}{E(Pb)_{1975}} \times \frac{S(D)}{S(Pb)} \times \frac{VMT_{1995}}{VMT_{1975}} \times C(Pb)_{1975}$$

Where:

$C(D)_{1995}$ = Projected ambient concentration of diesel particulate (ug/m^3)

$E(D)_{1995}$ = Fleet-average diesel particulate emission factor in 1995 (g/mi)

$E(Pb)_{1975}$ = Fleet-average emission factor for lead in 1975 (g/mi)

$S(D)$ = Dispersion factor for diesel particulate emissions

$S(Pb)$ = Dispersion factor for lead emissions

VMT_x = Total urban vehicle miles travelled in year x

$C(Pb)_{1975}$ = Urban ambient lead concentrations in 1975 (ug/m^3)

A fleet-wide lead emission factor for 1975 was calculated using formulas and tables developed in a recent EPA report.[8] Vehicles from 1956 to 1975 were considered, along with associated travel fractions and fuel economies. The calculations accounted for lead emissions from leaded, unleaded, and misfueled vehicles; for 1975, the national average lead contents for "leaded" and "unleaded" gasolines were 1.82 and 0.014 grams/gallon, respectively. Amount of consumed lead that is emitted with the exhaust was assumed to be 75 percent for leaded-fueled vehicles and 30 percent for vehicles designed to use unleaded fuel. Separate emission factors were calculated for each of five vehicle classes (LDV, LDT1, LDT2, HDT1, and HDT2), and then were weighted according to their relative urban VMT's to arrive at a fleet-wide average lead emission factor for 1975. The final calculated value was 0.1265 grams of elemental lead per vehicle mile travelled.

Dispersion factors for diesel and lead particulate matter were determined through a review of available data and literature. It was generally concluded in all examined studies that diesel particles are relatively small in diameter, compared to particulate matter produced from leaded gasoline.[8-11]

The majority of the diesel particles are found to fall within the "accumulation mode" (0.023 to 1.0 um) [10], with the median diameter between 0.10 and 0.25 um.[9,10] Due to its small size, diesel particulate disperses similar to a gas; therefore, S(D) equals 1.0.

Arriving at a lead dispersion factor for the model involved a review of findings from studies using various methods of analysis.[1-5,12-17] One method of estimating lead dispersion is based upon the particle size distributions of lead in both vehicle exhaust and ambient air samples. Before comparing the two distributions, a "cut-off" diameter of one um was chosen; Cantwell, et. al.,[14] states that "particles of less than one micron in diameter...(will) become airborne and remain suspended for a significant length of time." Huntzicker[4] estimates that approximately 25 percent of all exhausted lead particles are less than one micron in diameter, while about 64 percent of all airborne particles fall into this range. Assuming that all particles with diameters less than one micron will become airborne, a lead dispersion factor of 0.39 was calculated (0.25/0.64). In other words, an estimated 39 percent of all exhausted lead particles will become airborne, based on a particle size "cut-off" of one micron.

Another method of estimating S(Pb) in the model is to relate emission and ambient levels of lead and other vehicle-related pollutants, such as CO and EC (elemental carbon). Based on calculated 1980 emission factors and ambient concentrations for Pb and CO, dispersion of lead is estimated at values between 33 and 44 percent.[1,4] Assuming EC disperses much like a gas, Cass'[5] ratios of EC/Pb for "highway signature" versus ambient conditions can be used to calculate a 57 percent lead dispersion factor.

Other methods of estimating lead dispersion include soil analyses and actual measurements in enclosed areas. Ward[17] found that approximately 42 percent of exhausted lead remained airborne; he accounted for deposited lead in analyses of soil and vegetation within 250 m of a low-traffic state highway. Cantwell, et. al.,[14] reports the results of a study where vehicles were driven back and forth inside a sealed tunnel; when exhaust emissions were compared to ambient concentrations inside the tunnel, it was found that 46 percent of the exhausted lead particles remained airborne.

Averaging the results of these various studies, one finds that approximately 43 percent of all exhausted lead particles become airborne; therefore, a value of 0.43 is used for S(Pb) in the model equation.

It should be noted that only lead monitors in areas of no known large stationary sources of lead were chosen for this analysis; the majority of the non-automotive sources of lead emissions reside in a few identifiable areas which have been excluded from this analysis. For these reasons, it can be assumed that 100 percent of ambient lead can be attributed to mobile sources (in the context of this study), despite the fact that, nationwide, mobile sources only account for approximately 88 percent of all lead emissions.[12]

The nationwide VMT and urban estimates presented in Table 3-1 show urban VMT growth to be 40 percent between 1980-95. Since the Energy and Environmental Analysis projections[18] do not go back to 1975, a second DOE-sponsored study was used to derive the urban VMT growth between 1975-80. This Oak Ridge National Laboratory study estimated VMT growth to be 14 percent between 1975-80.[19] Combining these two figures yields an overall VMT growth between 1975-95 of 60 percent.

The use of the factors mentioned above results in the following general equation, which can be used to convert the monitored ambient lead concentrations into estimates of urban diesel particulate concentrations:

$$C(D)_{1995} = \frac{E(D)_{1995}}{0.1265 \frac{\text{grams lead}}{\text{mile}}} \times \frac{1.0}{0.43} \times 1.60 \times C(Pb)_{1975}$$

or

$$C(D)_{1995} = 29.4\ E(D)_{1995} \times C(Pb)_{1975}$$

Since each of the four scenarios in this analysis has a specific average diesel particulate emission factor (E(D)) associated with it, four discrete conversion factors are produced relating urban ambient concentrations of lead to diesel particulate levels. As an example, using the fleet-wide urban diesel particulate emission factor for best estimate sales and the relaxed scenario from Table 3-2 results in a factor of 2.62. This means that 1995 urban diesel particulate concentrations are projected to be 2.62 times larger than 1975 urban lead levels.

Table 3-3 presents the lead-based estimates of diesel particulate concentrations for each scenario for 28 cities included in the National Air Surveillance Network (NASN) for lead in 1975. These monitor stations were selected from a larger lead data base as they were known to be in areas having no large stationary sources of lead emissions, and to be above 12 meters in height in order to best represent large scale average urban lead concentrations. Table 3-4 presents the range of concentrations of diesel particulate for each scenario as a function of city size.

For the best estimate sales and relaxed particulate standards scenario, the ambient air diesel particulate concentrations range from a low of 1.2 ug/m^3 for the city of Kansas City, Kansas to a high of 7.1 ug/m^3 in Los Angeles. The other scenarios show similar ranges with the highest projected concentration occurring in the worst case sales, relaxed standards scenario, as expected (10.5 ug/m^3). In comparing the best estimate sales scenarios it can be seen that the base scenario will result in an estimated 38 percent reduction in the 1995 ambient diesel particulate concentrations compared to the relaxed scenario. This could constitute as much as a 2.6 ug/m^3 reduction (Los Angeles) or as little as a 0.5 ug/m^3 reduction (Kansas City, Kansas) in diesel particulate levels.

This same methodology can also be applied to 1980 diesel particulate emissions to show the change in estimated ambient diesel particulates between 1980-95. Using the 1980 diesel particulate emission factors from Table 3-2 and a VMT growth

Table 3-3

Ambient Diesel Particulate Concentrations Based on Lead Surrogate Model (ug/m^3)

		1995			
		Best Estimate Sales		Worst Case Sales	
City	1980	Relaxed	Base	Relaxed	Base
Population Greater Than 1,000,000					
Houston	2.5	5.5	3.4	8.1	5.4
Los Angeles	3.0	7.1	4.4	10.5	6.8
New York	1.2	2.8	1.7	4.1	2.7
Philadelphia	1.5	3.5	2.1	5.2	3.4
	1.4	3.2	2.0	4.8	3.1
Population Between 500,000 and 1,000,000					
Boston	1.1	2.5	1.5	3.6	2.4
Denver	1.1	2.5	1.5	3.7	2.5
Kansas City	0.9	2.1	1.3	3.1	2.0
New Orleans	1.3	2.8	1.7	4.2	2.7
Phoenix	2.5	5.5	3.4	8.1	5.4
Pittsburgh	0.9	2.2	1.4	3.3	2.1
San Diego	1.3	3.0	1.8	4.4	2.9
St. Louis	1.4	3.1	1.9	4.6	3.0
Population Between 250,000 and 500,000					
Atlanta	1.2	2.8	1.7	4.1	2.7
Birmingham	1.4	3.2	2.0	4.7	3.1
Cincinnati	0.9	2.1	1.3	3.2	2.0
Jersey City	1.2	2.7	1.7	4.1	2.6
Louisville	1.2	2.6	1.6	3.7	2.5
Oklahoma City	1.9	4.4	2.7	6.4	4.2
	1.2	2.7	1.7	4.0	2.6
Portland	0.9	2.1	1.3	3.2	2.0
Tucson	0.8	1.9	1.2	2.9	1.9
Yonkers	1.4	3.0	1.9	4.5	3.0
Population Between 100,000 and 250,000					
Kansas City, KA	0.7	1.6	1.0	2.4	1.5
	0.5	1.2	0.8	1.7	1.1
Mobile	1.2	2.6	1.6	3.7	2.5
New Haven	1.3	3.0	1.9	4.5	2.9
Salt Lake City	1.2	2.6	1.6	3.9	2.5
Spokane	0.7	1.5	1.0	2.2	1.5
Trenton	1.1	2.4	1.4	3.4	2.2
Waterbury	2.2	4.9	3.1	7.3	4.8

Table 3-4

Average Ambient Diesel Particulate Concentrations by City Population (ug/m ^{3})

City Size Grouping (Population)	1980	1995			
		Best Estimate Sales		Worst Case Sales	
		Relaxed	Base	Relaxed	Base
Greater than 1,000,000	1.2-2.7	2.6-6.2	1.6-3.9	3.9-9.2	2.6-6.0
500,000-1,000,000	0.8-1.8	1.8-4.1	1.2-2.5	2.8-6.0	1.8-3.9
250,000-500,000	0.9-1.5	2.0-3.4	1.3-2.1	3.1-5.1	2.0-3.3
100,000-250,000	0.6-1.6	1.3-3.6	0.7-2.2	1.8-5.4	1.2-3.5

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* Ranges are average values plus and minus one standard deviation.

rate of 14 percent (as indicated previously for 1975-80), a conversion factor of 1.16 is calculated; this factor, multiplied by the individual ambient lead concentrations, yields the 1980 estimates of urban diesel particulate concentrations found in Tables 3 and 4. As can be seen, ambient diesel particulate levels in 1995 will increase over 1980 levels under all scenarios. For example, between 1980-95, diesel particulate concentrations in Los Angeles would increase by 4.1 ug/m^3 under best estimate sales and the relaxed scenario, versus 1.4 ug/m^3 under best estimate sales and the base scenario.

Another characteristic difference between the present year (1980) urban ambient diesel particulate projection and the 1995 projections are that the proportion of LDDs versus HDDs and hence their impact on air quality are substantially different. LDDs produce only 12 percent of total diesel particulate emissions in 1980, but between 36 and 61 percent in 1995, depending on which scenario is chosen. For a city such as Los Angeles, this translates to an increase in urban ambient LDD particulate concentrations of 2.1 ug/m^3 and 2.2 ug/m^3 for base and relaxed standards with best estimate sales, respectively. Of course, by analogy the impact of heavy diesel vehicle categories (MDV/LHDV and HHDV) on urban air quality (1995 versus 1980) is proportionately less than the overall fleet, though in absolute terms still increasing.

Prior analyses have been performed by EPA on the impact of diesel particulate on urban air quality. The most pertinent studies are those done for the regulatory analyses for the light-duty diesel particulate standards and the HDD particulate standards.[20,21] In both of these regulatory analyses an identical lead-based air quality projection was made for diesel particulate. The only differences between these projections and the present study would be the diesel particulate and lead emission factors, lead dispersion factor, and the year for which the projections were made. The more recent heavy-duty analysis will be used as the primary comparison to the present study.

The analysis of the urban air quality impact resulting from diesel particulates, as calculated in this report, is approximately 45 percent lower than the previous analysis based on a comparison between the midpoint of the previous range of ambient concentrations and the best estimate relaxed scenario in this analysis. For example, the urban ambient level of diesel particulate in this study was estimated to be 7.1 ug/m^3 in Los Angeles under the best estimate relaxed scenario while the corresponding value in the previous analysis for the uncontrolled fleet was approximately 12.9 ug/m^3. The primary reason for this difference is the fact that non-trap emission levels for LDDs are well below those projected three years ago and that the relaxed-scenario standard for HDDs includes a slight degree (15 percent) of control.

IV. Microscale Air Quality Analysis

Certain very specific localized areas are known to be affected by motor vehicle emissions to a greater extent than urban areas as a whole (and the locations of the lead monitors). Among these localized areas (hereafter called microscale areas) are urban expressways, street canyons, roadway tunnels, parking garages and residential garages. In a previous effort by EPA designed to evaluate potential hazards due to unregulated pollutants emitted from motor vehicles, a set of ambient air dispersion models and model parameters were developed and validated.[22]

These models, while mathematical in nature, were validated based on known concentrations of CO in these microscale areas. As such, these models can be considered reasonably accurate for the specific geographical and meteorological situations being examined. However, as the relationship between diesel particulate and CO emission factors may differ under specific conditions, these models can only be considered to be good estimates when applied to diesel particulate modeling. However, this approach is perhaps the best assessment available for localized estimates of diesel particulate concentrations.

The aforementioned work identified a set of typical and severe situations for each of the microscale areas, differing by vehicle traffic volume, windspeed, and other factors influencing ambient concentrations. The results of the earlier work allow calculation of the ambient air concentration for any of these microscale areas (in either the typical or severe situations) based only on the pollutant emission factor. If a pollutant is assumed to be evenly distributed within the microscale and is of low short-term reactivity, then the pollutant emission factor is multiplied by the conversion factor (one for each microscale area situation) to obtain an estimate of the ambient concentration at the specific microscale location.

Table 3-5 presents the various selected microscale areas and their corresponding conversion factors. These factors represent the ambient concentrations of any applicable pollutant (based on above criteria) estimated to occur in each of these microscale areas for a vehicle emission factor of 1 g/mi. These conversion factors can not be directly applied to lead emissions due to uneven distribution of lead throughout the microscale (deposition/dispersion charateristics). However, CO, which is 100 percent dispersed, would be an applicable pollutant in this case.

Table 3-5

Summary of Microscale Situation Concentrations

Situation	Microscale Conversion Factor (ug/m^3 per g/mi)
1. Roadway Tunnel	
Typical - Lowry Hill, Minnesota	1,123
Severe - Baltimore Harbor Tunnel	2,856
2. Street Canyon (sidewalk receptor)	
Typical - 4 lane canyon, 800 vehicles/hr., 8 mph windspeed	42
Severe - 6 lane canyon, 2400 vehicles/hr., 2 mph windspeed	282
3. On Expressway (Wind: 315 deg. relative, 2.2 mph)	
Typical - San Antonio I-410,	124
Severe - Los Angeles I-10,	506
4. Beside Expressway	
100 meters away-downwind	105

The particular values listed in Table 3-5 for the typical situations were selected to be reasonably representative of the desired types of areas. The concentrations represented by the severe situation for each scenario would be expected to occur only a small percentage (1 percent) of the time on a nationwide basis. However, in a given specific area, the severe case could occur much more frequently. For example, the severe expressway situation used a segment of the Santa Monica freeway in Los Angeles, which is a 10-lane freeway with a 200,000 vehicle/day traffic count. The windspeed and direction were typical of this location. While this kind of traffic flow is severe for most urban expressways in the nation (impossible for most), it is a definite regular occurrence for this expressway and other busy large expressways in large metropolitan areas. Thus, while the severe situation would not be expected to occur frequently on urban expressways in general, there is a real possibility of frequent occurrence in the few very busy freeway segments in large cities.

Table 3-6 presents the results of the microscale area calculations for the four scenarios. The range of localized diesel particulate concentrations in Table 3-6 constitute an estimate of the levels which might be expected in these areas in 1980 and 1995. These levels are not to be construed as anything like average urban levels or average personal exposures. In fact, the overall population exposure contributed by these very high, short-term levels is probably relatively small. However, to the extent that the population is exposed as they pass through these microscale areas in their day to day activities, these localized area diesel particulate concentrations could constitute an impact on their health or welfare.

For example, high localized concentrations of diesel particulate on an expressway (30-70 ug/m^3 in 1995) may be reflected in reduced short-term visibility or increased short-term odor which may impact on the health and welfare of the commuting public. However, current levels of diesel particulate in an identical situation could already be approximately 25 ug/m^3.

An examination of Table 3-6 shows the wide variety in potential localized area concentrations of diesel particulate. These projected levels range from as low as 2 ug/m^3 for a typical street canyon under the best estimate sales and base standards scenario to as high as 378 ug/m^3 for a severe roadway tunnel under the worst case sales and relaxed standards scenario. The lowest levels of this range roughly correspond to the overall urban area concentrations presented in Table 3-3. This finding is consistent with the fact that some of the

Table 3-6

Microscale Diesel Particulate Concentrations (ug/m³)

		1995			
		Best Estimate Sales		Worst Case Sales	
	1980	Relaxed	Base	Relaxed	Base
I. Roadway Tunnel					
Typical	57	100	62	149	97
Severe	145	254	158	378	246
II. Street Canyon					
Typical	2	4	2	6	4
Severe	14	25	16	37	24
III. On Expressway					
Typical	6	11	7	16	11
Severe	26	45	28	67	44
IV. Beside Expressway	5	9	6	14	9

fixed site monitors used in the lead ambient monitor network are sited in locations such as street canyons (on top of tall buildings) or near expressways and concentrations in these localized areas, under typical conditions, may approach the overall urban area averages.

An analysis of the overall differences between the relaxed and base scenarios yields the same percentage differences as those found for urban emissions in general in Chapter 2. These differences can be translated to an increase in localized diesel particulate concentrations of from as low as 2 ug/m^3 for a typical street canyon situation to as high as 96 ug/m^3 for a severe roadway tunnel situation. Comparing the increases in these projected 1995 localized diesel particulate concentrations to the concentrations which may be occurring now (1980) results in the observation that, for the severe roadway tunnel situation, present levels of diesel particulate may be expected to be on the order of 145 ug/m^3, which can be compared to the projection for the best estimate sales relaxed control scenario in 1995 of 254 ug/m^3, or the projection for the best estimate sales base control scenario of 158 ug/m^3.

V. Population Exposure Analysis

A. Introduction

The population exposure estimates used in this report are based on a general air pollutant exposure model, called the NAAQS Exposure Model (NEM), developed by OAQPS for the evaluation of relative population exposures under alternative NAAQS [23]. The NEM is an activity pattern model that simulates a set of population groups called cohorts as they go about their day-to-day activities. Each of these cohorts are assigned to a specific location type during each hour of the day. Each of several specific location types in the urban area are assigned a particular air quality value based on fixed site monitor data. The model computes the hourly exposures for each cohort and then sums these values over the desired averaging time to arrive at average population exposures and exposure distributions. Thus, the model simulates pollutant concentrations in urban areas by relating these concentrations to fixed-site monitor levels; in turn, the model simulates the activities of people by relating the population to a fixed set of cohorts with defined activity patterns.

For example, a certain fraction of the total urban population might be assigned to an office worker cohort with a home-work-home activity pattern. This cohort would experience a consistent set of microenvironments, such as home, transportation, and office, in a normal day's activity. Each of these microenvironments would have an associated pollutant concentration related to the fixed-site monitor level for the specific time of day and date. The fixed-site monitor levels are adjusted to correct for the differences that typically exist between the monitored locations and the microenvironment location. These adjustments are general enough to account for multiplicative and additive types of correlations between the monitors and locations.

A unique feature of the model is that it separates concentrations, people, places, and time (all of the important ingredients of exposure) into discrete elements. The concentrations are broken up into values determined by the precision of the fixed site monitors (e.g., for CO into whole integers of ppm). The people or urban population of an area are separated into cohorts that are assumed to have specific activity patterns and related exposures. The places are separated into a discrete number of areas which are assumed to have identical pollutant concentrations over a given period of time. Thus, the definition of places may be influenced by the type of pollutant studied and its emission sources. Time is separated into the smallest unit of measure which is desired. Since this methodology was designed to be used with the NAAQS, which are based, at a minimum, on a 1-hour time period, one hour is the shortest time period considered by the general model. Longer averaging times, such as 24 hours or a year, can be obtained by calculating the appropriate averages from the 1-hour exposures.

The general NEM approach has been used by OAQPS to develop specific models for a number of criteria pollutants, such as CO, SO_2, NO_2 and particulate.[24] These pollutants have been studied by applying the specific pollutant data base from the appropriate EPA monitoring program and by designing the place designations to those most appropriate to the pollutant. Place or location designations in the NEM are determined as exposure districts or exposure neighborhoods, depending on whether the pollutant of interest is a point or dispersed source of emissions, respectively. The exposure district approach is more geographical in nature and relies on the fact that pollutants which are primarily emitted by large point sources can be adequately characterized by exposure districts of fairly large areas. In contrast, pollutants with emission sources which are dispersed throughout an urban area (including mobile sources) are best characterized by considering exposure neighborhoods with common exposure patterns. These neighborhoods may be spread out in a random fashion throughout the urban area.

The general NEM modeling approach using monitoring data in four cities (Chicago, Los Angeles, Philadelphia, and St. Louis) was applied to the criteria pollutants mentioned above resulting in four average exposures and exposure distributions. A limited set of 24 total monitors in four cities was used because of the extensive computer time required to run the NEM. A rough nationwide exposure extrapolation has been developed by OAQPS, which involves relating each of the large urban areas in the country to the most similar of the four modeled NEM cities.[24]

Direct measurements of diesel particulate levels in overall urban areas have not been made because of the difficulty in distinguishing diesel particulate from other carbonaceous particulate. The use of a surrogate approach to relate diesel particulate to some other pollutant which can be readily measured in urban areas appears to afford the best chance of obtaining reasonable estimates of diesel particulate concentrations, as was the case for urban air quality estimates in the previous section of this report. Of the criteria pollutants which have been assessed by OAQPS using the NEM methodology, CO appears to have the most desirable characteristics as a surrogate for diesel particulate. Atmospheric CO concentrations are generally recognized as being essentially totally attributable to mobile sources, whereas other automotive pollutants are also emitted by stationary sources. The dispersion characteristics of CO are also very similar to those of diesel particulate matter, since diesel particles are found to be relatively fine and are, therefore, assumed to disperse essentially like a gas.

The CO NEM is designed to provide an overall estimate of CO population exposure related to different values of the CO NAAQS. While the selection of CO and the NEM methodology is perhaps the best basis for a mobile source assessment of population exposure, a number of modifications to the NEM CO assessment are necessary and/or desirable in order to provide the best estimate of diesel particulate exposure.

One modification to the standard NEM methodology involves the removal of all indoor sources of CO, such as gas stoves, from the air quality inventories. This change allows for a more precise estimate of automotive sources, particularly indoors, where the contribution of outdoor emissions is still present without the confounding presence of indoor sources. This is the most important modificaton to the model in order to allow a reasonable estimate of automotive exposure to CO and, via an appropriate conversion, to diesel particulate.

Other desirable modifications to the NEM, which are planned for the near future, include an effort to correct the model for a suspected underestimation of mobile source microscale area contributions, and an effort to design a national extrapolation procedure expressly for mobile sources. A modified version of the NEM methodology is expected to produce slightly higher exposure levels than the original version; however, it is felt that the current version is adequate for the purposes of this document.

The NEM-based exposure estimation methodology used in this report provides both an average CO exposure and CO exposure distribution for the four cities in the data base. The average CO exposure results are used to develop the nationwide exposure estimates for diesel particulate in 1995. The exposure distribution form of the methodology is not essential for the uses of this report and will not be presented here. However, for the sake of completeness and because the distributions do present information on exposure ranges of diesel particulate which may be interesting in placing the exposures in perspective, the exposure distributions for diesel particulate will be presented later in this chapter.

B. Past Exposure Efforts

Before discussing the details of the diesel particulate exposure estimate derived in this report, it may be useful at this point to compare the NEM methodology to the general methodologies used in previous EPA assessments of mobile source pollutant exposures. Two different assessments have been used in the past: 1) one based on a methodology by Pedco for a previous EPA diesel particulate risk assessment and, 2) one based on a methodology by SRI for an EPA benzene risk assessment.

The Pedco exposure assessment used an Air Quality Display Modeling (AQDM) approach wherein the urban area to be modelled was broken up into a set of geographical grids where the grid population and grid pollutant concentrations were combined into an exposure for each grid.[25] The Pedco approach used TSP to derive the original predicted concentrations of particulate and adjusted these predictions based on TSP monitor levels. However, since the emission pattern and ambient distribution of TSP may be very different from diesel particulate due to the large contribution of non-mobile sources to TSP emissions, this is and was thought to be a source of possible error in the Pedco assessment. Also, no effort was made to simulate different activity patterns as was done by the NEM model. The Pedco approach was applied to only one city, Kansas City, and this single result was extrapolated nationwide. The Pedco approach was very valuable at the time of its development as a coarse, but usable, first estimate of the population exposure to diesel particulate; however, the present NEM model is felt to be a more detailed, precise approach and probably, therefore, yields a more accurate result. It is not possible to directly compare the results of these two approaches (NEM and Pedco) because of the different emission factors used. However, it is estimated that the current NEM assessment results in exposures which are roughly a factor of 10 higher than those in the Pedco report. At the time of the preparation of the Pedco report, and its subsequent use in the EPA's preliminary risk assessment for diesels, it was thought that the Pedco assessment might be low, primarily because Kansas City was not thought to be the most typical urban area with respect to automobile emissions. Thus, while this factor is significant, it is not unexpected or unusual in our view, but rather indicates that the current NEM approach represents a more correct and precise exposure assessment for mobile sources.

The SRI modeling approach used for the EPA benzene risk assessment estimated the mobile source contribution using an area wide dispersion model called the Hanna-Gifford dispersion model.[26] This approach is very simplistic, requiring only an estimate of an urban area's vehicle registrations, VMT, area size, average annual wind speed, and vehicle emissions. A relatively limited examination of population activity patterns was used by SRI to estimate the influence of the many different sources of benzene, but only the area-wide dispersion-based averages were used for the automobile contributions. Comparison of the exposures calculated by the SRI benzene assessment to the NEM exposures results in the finding that the NEM based exposures are roughly a factor of five higher than the SRI estimates for a comparable emission factor. The SRI report states that the automobile contribution to the benzene assessment are probably underestimated because of the fact that the area-wide model used may not adequately reflect the high localized concentrations believed to occur around automobile-use areas.[26] The relatively close agreement between the SRI and NEM exposure assessments and the intuitive logic discussed earlier on why the NEM should be higher leads to the conclusion the NEM model used in this report is probably the most valid approach currently available.

C. Average Nationwide Diesel Particulate Exposures

Table 3-7 presents the NEM based average CO concentrations for the four cities used in NEM program. The average CO exposure concentrations for each city in Table 3-7 are combined in order to provide the desired nationwide average exposure. A simple method to use, and the one used by both the CO NEM and

Table 3-7

Average Total CO Exposure in Four Cities

City	CO ppm (Annual avg.)	Population of City As Used in CO NEM	Associated Total Urban Population In Cities 200,000 1970
Chicago	1.8 ppm	2,363,014	38,894,365
Los Angeles	3.0	7,716,895	26,339,249
Philadelphia	1.3	2,933,790	10,553,523
St. Louis	2.0	1,221,461	17,350,712
Overall	2.12		93,137,849

this report, groups each of the large urban areas in the country (populations greater than 200,000) with one of the four modeled cities according to overall urban characteristics, including populations, vehicle-use patterns, etc.[24] Under this nationwide extrapolation, a large portion of the population (43 percent) is grouped under Chicago. While this nationwide extrapolation seems reasonable and valid as an estimate, it is most likely the least precise part of this assessment. Thus, it is one of the areas that ECTD is continuing to investigate as part of the ongoing mobile source exposure estimate project. ECTD intends to use a large bank of CO monitor data, perhaps from the EPA SAROAD data base, selected with a view toward mobile source contributions, to provide an extrapolation to the nationwide situation.

The national population (Column 4) in Table 3-7, as mentioned previously, counts only persons in urban areas with populations greater than 200,000. While it is desirable to extend this analysis to the population in all urban areas, including those with populations less than 200,000, the likelihood that the exposures in smaller urban areas would be lower than any of the four NEM cities prevents this from being precisely done. Therefore, we have limited our analysis to the populations in the large urban areas without considering the exposures of rural or small urban areas.

The aforementioned nationwide extrapolation to the NEM-CO average output results in the calculation of an overall average nationwide concentration (based on CO) of approximately 2.1 ppm. This total adjusted national average (2.1 ppm) can then be manipulated into a diesel particulate national average by ratioing CO and diesel particulate emission factors, and then multiplying the result by 2.12 ppm. The national average CO emission factor for 1978 (the same year as the CO NEM data base) is estimated to be 62.3 g/mi.[27] However, since future year VMT is expected to increase by about 45 percent, the diesel particulate emission factor should be adjusted upward by a factor of 1.45 for 1995.[18] The 1995 diesel particulate emission factors are the same as those used in the air quality analyses (see Table 3-2).

Table 3-8 presents national average diesel particulate concentrations for each of the four main scenarios. These values will be used in Chapter 5 to estimate the diesel particulate cancer risk.

The total population exposure (from Table 3-8) for the best estimate sales and relaxed standards scenario is estimated to be 61 percent higher than the exposures calculated for the corresponding base case standards. However, separating the

Table 3-8

Total National Diesel Particulate Exposure in 1995

	Annual Average Diesel Particulate Exposure (ug/m^3)			
	Best Estimate Sales		Worst Case Sales	
	Relaxed	Base	Relaxed	Base
LDV	1.23	0.92	2.73	2.07
LDT	0.60	0.64	0.92	0.87
MDV/LHDV	0.54	0.27	1.07	0.52
HHDV	2.61	1.34	2.66	1.37
Total	4.98	3.09	7.40	4.82

light- and heavy-duty components of these exposures, individual contributions to this increase in exposure with relaxed standards are 14 percent for LDVs (LDVs and LDTs) and 86 percent for HDVs (HDV/LHDV and HHDV). These data can be interpreted as meaning that the bulk of the increase in exposure with best estimate sales and relaxed standards can be attributed to HDVs with a comparatively small contribution from LDVs.

If the worst case sales projections are used to derive relationships between the relaxed and base scenarios above, then the overall population exposure is increased 54 percent with LDDs contributing 28 percent of the increase, and HDVs 27 percent.

D. Exposure Distribution for Diesel Particulate

In addition to the national average exposure derived in the previous section, this mobile source model can be used to identify a distribution of exposures among discrete concentration ranges. A manipulation of this information in a manner analagous to the previous discussed average exposure can be used to provide a national average diesel particulate exposure distribution. For convenience this distribution is presented in Table 3-9 as a range of percentages for the two cities with the lowest and highest exposures versus the diesel particulate concentration range (dependent on the scenario).

The exposure distributions included in Table 3-9 can be used as a relative illustration of how the total exposure is broken down into concentration ranges. While this data is not used further in this analysis, in the event that a non-linear risk model is used in the future to estimate diesel cancer risk, data such as those in Table 3-9 will be necessary for estimating the cancer risk to individuals.

A brief inspection of the data in Table 3-9 show that while there are distinct differences between each city's exposure distribution, there is the common feature wherein most of the diesel particulate exposures (95-99 percent) are in the lowest range. The corresponding ranges of diesel particulate exposure are from 0-10 to 0-21 ug/m^3 depending on scenario. If future interest is generated on this kind of exposure index, a way to further break out the exposures within this lowest range will be necessary and this effort is underway as part of the ongoing ECTD project on developing a mobile source exposure assessment methodology.

Table 3-9

Diesel Particulate Exposure Distribution

Diesel Particulate (ug/m^3)				Range of Population Exposed %	
Best Estimate Sales		Worst Case Sales		High	Low
Relaxed	Base	Relaxed	Base	Los Angeles	Philadelphia
117-	73-	174-	114-	0.000654	0.000250
106-117	66-73	174-174	102-114	0.000345	0.000000
94-106	58-66	139-157	91-102	0.000941	0.000263
82-94	51-58	122-139	79-91	0.006804	0.001121
70-82	44-51	105-122	68-79	0.008965	0.000555
59-70	36-44	87-105	57-68	0.036916	0.007867
47-59	29-36	70-87	45-57	0.066520	0.009422
35-47	22-29	52-70	34-45	0.185011	0.020867
28-35	17-22	42-52	27-34	0.552863	0.084505
21-28	13-17	31-42	20-27	1.292952	0.194798
16-21	10-13	21-31	16-20	2.825991	0.414451
0-16	0-10	0-21	0-16	95.022026	99.265896

VI. Support for Air Quality Analysis Used in This Chapter

For purposes of validation, the following sections will lend support to the basic theories behind the air quality analysis performed in this chapter. The first section contains a review of available historical data on mobile source emissions and ambient concentrations of both lead and carbon monoxide; the demonstrated similarity between trends in emissions and ambient concentrations offers support for the use of the "rollback" theory in projecting future ambient levels of both lead and CO. The second section contains a comparison of the air quality model described earlier in this chapter to other lead surrogate work contained in a recent EPA study. In this section, both models are used to calculate 1980 ambient diesel particulate concentrations for four U.S. cities; the similarity in results obtained with the different models lend support to their validity.

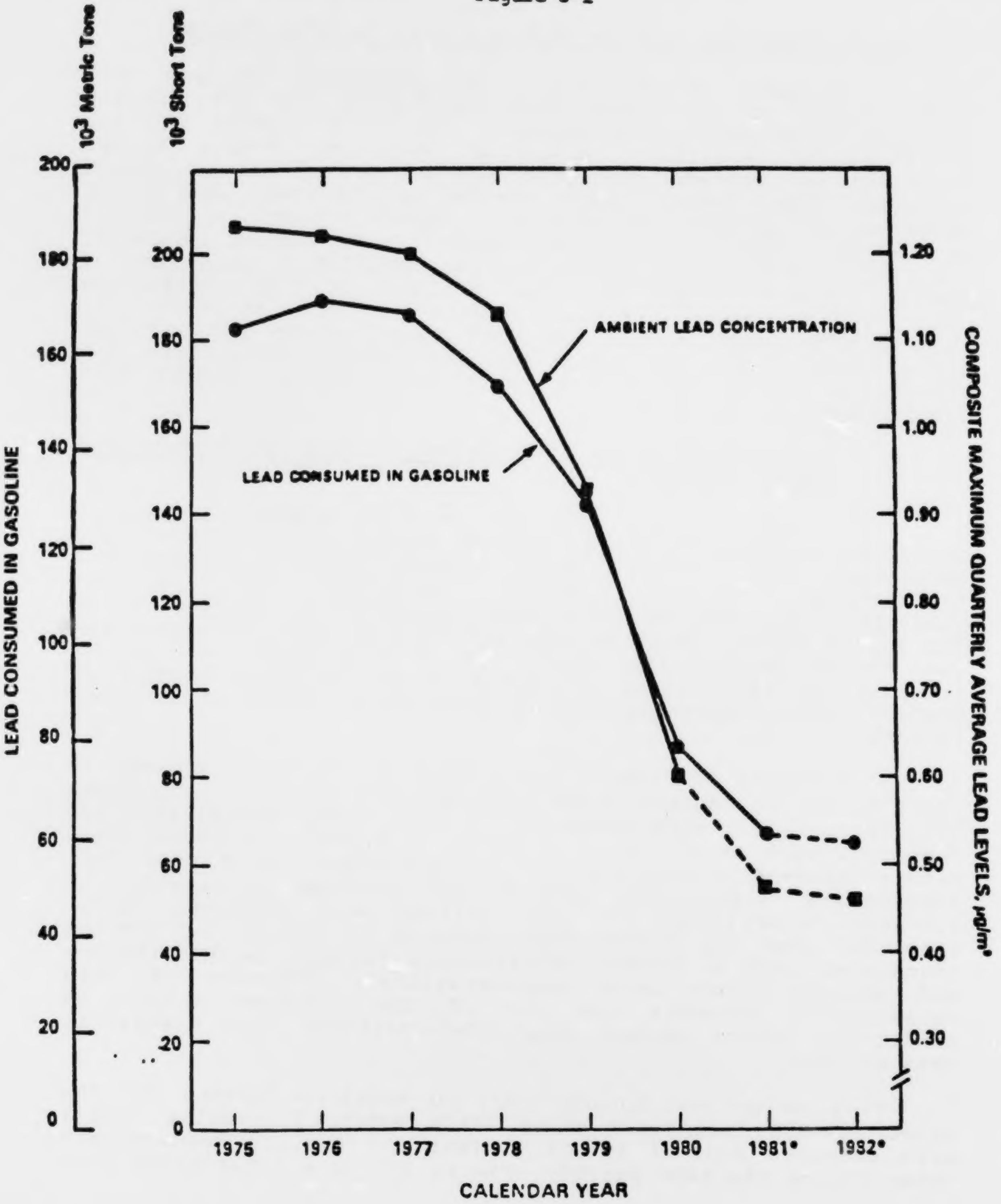
A. Validations of Rollback Theory as Applied to Lead and Carbon Monoxide

The theory behind the air quality models previously described is based primarily on the simple "rollback" modeling technique. The rollback model assumes that a proportional relationship exists between emissions and ambient concentrations, and uses this proportionality to project values for future years. Because lead and CO were used as surrogates for diesel particulate matter in this chapter, it is of interest to investigate the ability of rollback models to predict future concentrations of lead and CO based on emission factors.

In a recent EPA report [12], data on the lead consumed in gasoline and on ambient urban lead concentrations for the years 1975 through 1979 were examined. A very strong correlation was reported, with an r^2 value of 0.99. A graph of consumed lead versus ambient concentrations is presented in Figure 3-1. Because the relationship between lead consumed in gasoline is linearly proportional to lead emitted with exhaust, it is concluded that the correlation depicted in Figure 3-1 can be translated into a direct relationship between lead emissions and ambient urban lead concentrations. Evidence of this relationship supports the use of the rollback theory in predicting future ambient lead concentrations from historical data on lead.

In a second EPA report [28], CO emission factors for the years 1970-80 (estimated using EPA's MOBILE-2 computer model) were examined against actual ambient air quality measurements taken during the same period. Figure 3-2 is a logarithmic plot

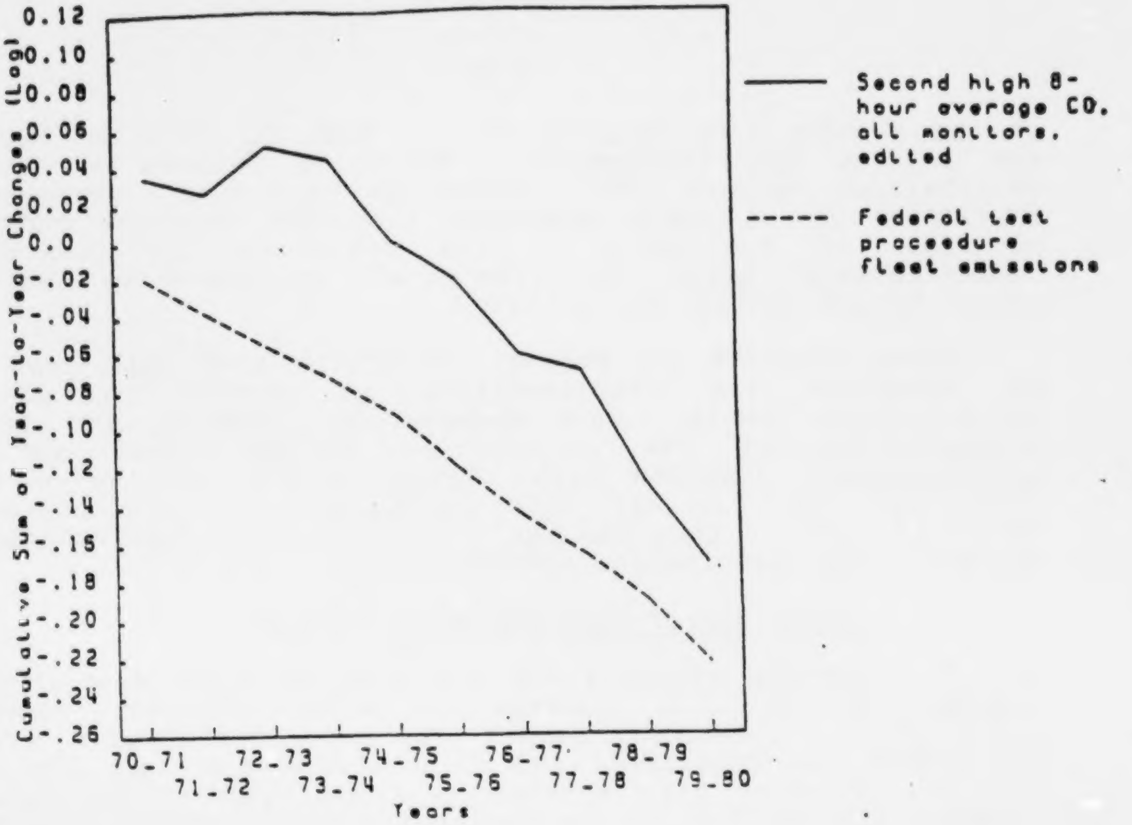
Figure 3-1



Lead consumed in gasoline (Du Pont, 1982) and ambient lead concentrations, 1975-1982. (Nunt and Neligan, 1982).

*DASHED LINES ARE ESTIMATES.

Figure 3-2


Cumulative Sum of Year-to-Year Changes (Log)
0.12
0.10
0.08
0.06
0.04
0.02
0.0
-.02
-.04
-.06
-.08
-.10
-.12
-.14
-.16
-.18
-.20
-.22
-.24
-.26
70_71
71_72
72_73
73_74
74_75
75_76
76_77
77_78
78_79
79_80
Years
Second high 8-hour average CO, all monitors, edited
Federal test procedure fleet emissions

of the changes from year-to-year in both CO emission factors and ambient CO measurements. The plot depicts a general correlation between the reductions in both parameters; therefore, it is again concluded that the rollback modeling technique is applicable to the projection of future CO concentrations based on historical relationships between emissions and ambient air quality.

These findings are strong indications that both lead and CO emissions are representative of trends in ambient concentration levels. This observation, coupled with earler documentation of the correlation between lead and CO concentrations (Section III), leads to the conclusion that either of the two are fair candidates for the surrogate pollutant used in this chapter's rollback model for estimating ambient diesel particulate concentrations.

C. Comparison of Lead Surrogate Models

This section compares the air quality model described in Section III of this chapter to other rollback modeling contained in a recent EPA study. The basic differences between the models are discussed, and calculations are made for four U.S. cities using both models. A comparison of the results supports the validity of the nationwide model developed in this chapter.

In a recent study performed by EPA's Office of Research and Development (ORD) in response to Section 214 of the Clean Air Act, lead was used as a surrogate in predicting ambient automotive particulate concentrations for several cities.[1] Like the lead surrogate model developed earlier in this chapter, ORD's models make use of the rollback theory; however, the models differ in that the ORD study examined automotive particulate concentrations as a whole, as opposed to just diesel particulate. Also, the ORD study used a baseline year of 1977 in its projections, while the model developed earlier in this chapter is based on 1975 ambient lead data.

Models developed in the ORD study[1] use a ratio of automotive particulate mass to lead mass in determining a lead emission factor for the baseline year (1977); this national average ratio for 1977 ranges between 5:1 and 8:1, and is based on emission data taken from ambient air monitoring sites. Using a 1977 fleet-average automotive particulate emission factor of 0.24 g/mi, the resulting lead emission factors for 1977 range between 0.030 and 0.048 g/mi. Because these emission factors are based on ambient data, as opposed to vehicle exhaust emissions, a lead dispersion factor is not necessary. (The model developed earlier in this chapter uses an exhausted lead emission factor of 0.1265 g/mi and a lead dispersion factor of 0.43; this translates into actual dispersed lead emissions of 0.054 g/mi for baseline year 1975.)

Ambient diesel particulate concentrations for four cities--St. Louis, Los Angeles, New York, and Kansas City (MO)--were calculated for 1980 using a modified version of the model developed in the ORD study. The model has the following basic form:

$$C(D)_{80} = \frac{E(D)_{80}}{E(Pb)_{77}} \times \frac{VMT_{80}}{VMT_{77}} \times C(Pb)_{77}$$

The 1977 lead emission factors are the same as those used in the ORD report (0.030 and 0.048 g/mi); the 1977 ambient lead concentrations used are also those reported in the ORD study.[1] The remaining parameters are those derived earlier in this chapter. A 1980 fleet-average diesel particulate emission factor of 0.0552 g/mi was used in the model equation (from Table 3-2). The VMT growth rate between 1977 and 1980 was estimated at 3.4 percent[19]; therefore, a factor of 1.034 was used as the VMT ratio in the equation. By substituting the above constants into the model equation, the 1980 ambient diesel particulate concentration can be estimated by a range of values; the lower and upper boundaries of the range are represented by multiplying the 1977 ambient lead concentration for each city by factors of 1.19 and 1.90, respectively.

The above calculations were performed for the four previously mentioned cities; Table 3-10 compares these values to those listed in Table 3-3 for each of the cities. For three of the four cities, the values previously calculated using the model developed in this chapter fall well within the ranges calculated with the modified ORD model. The concentration for the fourth city (Kansas City, MO) falls just slightly below the newly calculated range; this could be due to different base monitors being used in the two studies.

Overall, the comparison of results demonstrates that, although the basic input to the models may differ in form, the final concentrations arrived at are fairly consistent. This similarity in results, along with other documented support for the theories behind the surrogate approach, lends validity to the air quality model developed earlier in this chapter.

Table 3-10

Comparison of Ambient Concentrations Calculated With Two Different Models

City	1977 Ambient Lead Concentration (ug/m3)	Calculated 1980 Ambient Diesel Particulate Concentrations (ug/m3)	
		Modified ORD Model	Values from Table 3-3
St. Louis	0.89	1.1-1.7	1.4
Los Angeles	1.8-4.2	2.1-8.0	3.0
New York City	0.67	0.8-1.3	1.2
Kansas City, MO	0.9-1.4	1.1-2.7	0.9

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CHAPTER 4

VISIBILITY ASSESSMENT

I. Introduction

The most obvious effect of diesel particulate, especially in urban areas, is reduced visibility. In order to study this effect, there must be a means of measuring the relationship between diesel particulate and visibility levels. A method is needed to determine the visibility impact from a specific level of diesel particulate.

This chapter develops and applies a method for measuring the change in visibility caused by an increase in diesel particulate concentration. This is done on a city-by-city basis, yielding visibility levels for four regulatory scenarios.

No attempt is made in this study to estimate the dollar value of the projected changes in visibility. Such an analysis is beyond the scope of this report. For an economic analysis of the benefits of visibility improvements, the reader is referred to a recent study performed under contract to EPA[1] and other studies in the literature.

II. Modeling Visibility

There is no absolutely preferred method for modeling visibility; different measuring techniques are appropriate for various times and locations. The three types of visibility-related indices are: 1) direct measures of human perception, 2) measures of light intensities, and 3) measures of visual properties of air. Using observers to measure airport visual ranges is an example of direct human perception measurement. This is a subjective method which is difficult to convert to objective physical parameters. There exists no correlation between the methods of measuring direct human perception and diesel particulate. In measuring light intensities, the relationship between perceived contrast and measured physical contrast is also a subjective and complex one. (Contrast, combined color and brightness scales, and blue-red luminous ratios are examples of measures of intensities.) Because both of the above methods appear to be inadequate for relating the effect of particulate on visibility, the third method is, by necessity, the method of choice. The measuring of visual properties of air and airborne particles can directly relate the particulate matter from diesels to a reduction in visibility.

Visibility is defined as the greatest distance it is possible to see a prominent dark object against the sky at the horizon.[2] Middleton's Law[2] relates contrast and light intensity; both are reduced equally at horizontal views of objects against the horizon. Koschmieder's Law[2] goes a step further and relates visual range to the extinction coefficient. The typical observer can detect an object with 2 percent contrast against the background.[2] The mathematical formula describing Koschmieder's Law is:

$$L_v = \frac{3.91}{b_{ext}}$$

where L_v is the visual range, 3.91 is ln(.02) and b_{ext} is the total extinction coefficient.

Koschmieder's Law can be derived from the Beer-Lambert Law, which describes the more fundamental effect of the extinction coefficient (b_{ext}) on light intensity. The Beer-Lambert Law is:

$$I = I_o e^{-b_{ext}L}$$

Where:

I_o = light intensity at the object being observed, and

I = light intensity at a distance L from the object.

Described very simplistically, Koschmieder's Law simply states that objects become invisible when the ratio of I to I_o becomes 0.02. Substituting 0.02 for I/I_o into the Beer-Lambert Law and solving for L yields Koschmieder's Law.

As can be seen, the most important parameter in all of these laws is the extinction coefficient (b_{ext}). This coefficient is the sum of four components:

1. scattering by gas molecules, b_{Rg};
2. absorption by gas molecules, b_{ag};
3. scattering by particles, b_{sp};
4. absorption by particles, b_{ap}.

Diesel particulate impacts directly on the latter two processes. In order to gain insight into the relative role of diesel particulate in light attenuation, each of the four components of the extinction coefficient should be examined.[3]

A. Gas Scatter

The extinction coefficient due to scattering by gas molecules in the free atmosphere at sea level, known as Rayleigh scattering, is roughly 1.5 x 10^{-5} meters^{-1}; values of the extinction coefficient within a few percent of this have actually been measured.[2] If light degradation were due solely to gas molecule scattering, then the visibility would be approximately 260 kilometers by the Koschmieder formula. Thus, scattering by gas molecules does not play a major role in observed visibility degradation.

B. Gas Absorption

Nitrogen dioxide (NO_2) is the only absorbing gaseous specie present in high enough concentrations to have a significant effect on light absorption. In optics, NO_2 seems to be important only in plumes, not in the case of a well-mixed layer.[2] Therefore, absorption by gas molecules can be discounted in calculating the total extinction coefficient.

C. Particle Scatter

Particles with diameters in the range of 0.1 to 1.0 micrometer scatter light with the greatest effectiveness. Diesel particulate falls into this range. Figure 4-1 shows the ratio of mass to scatter coefficient as a function of particle radius. The duration of this scattering effect is prolonged for this size range, since such aerosols generally do not settle out by gravity and are not removed efficiently from the atmosphere except by incorporation into clouds and subsequent rainout. Studies show that they may persist in the atmosphere for several days.[4]

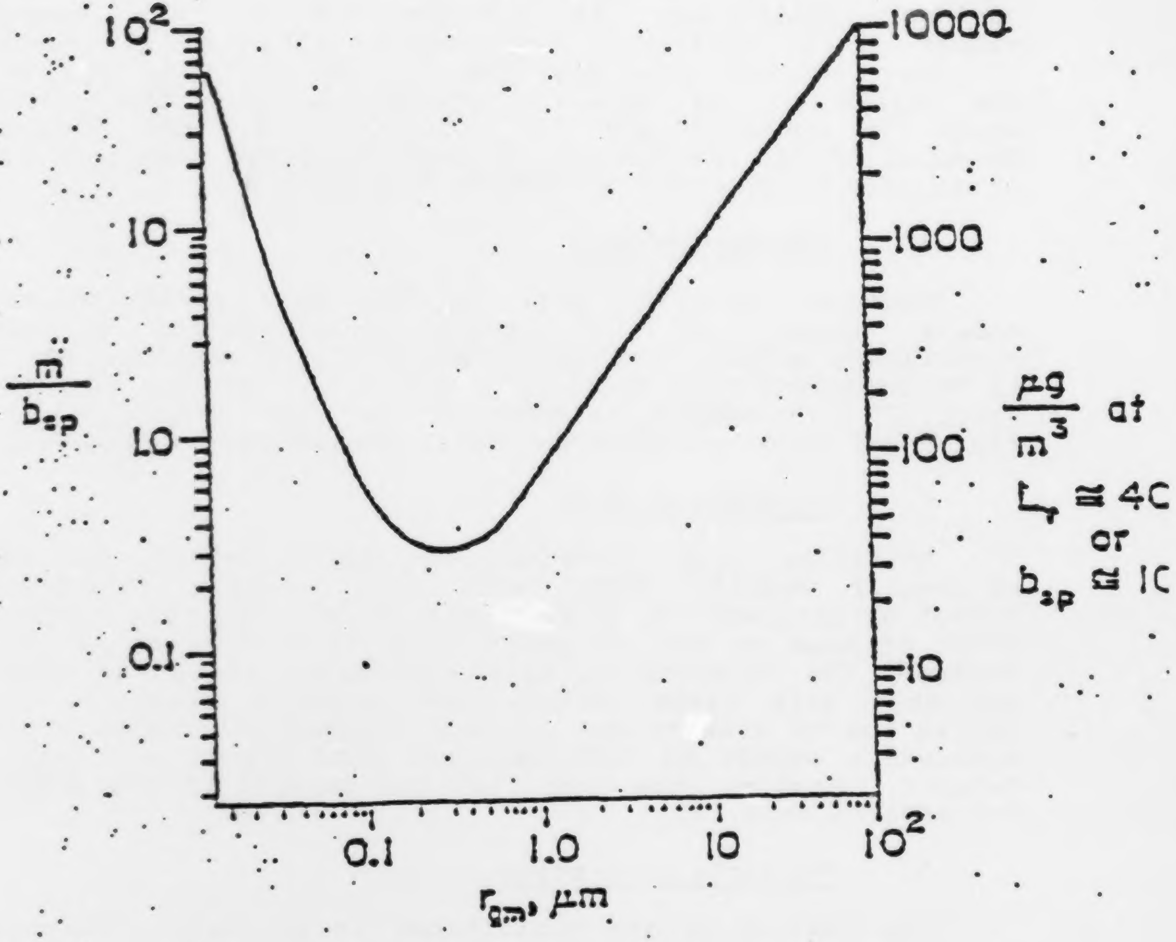
D. Particle Absorption

The most important contributor to particle absorption is graphite carbon; any sub-micron particles with a high carbon content will have a significant effect on visibility. Diesel particulate, with its 65-80 percent carbon content, falls into this category.[5]

III. Visibility Equations

Once all the factors involved are known, the computation of visibility levels due to a change in diesel particulate level is straightforward. According to the Koschmieder formula, the visual range is inversely proportional to the total extinction coefficient. The total extinction coefficient (b_{ext}) is the sum of the extinction coefficient for the base line visibility (b_o) and the extinction coefficients due to absorption and scattering of diesel particulate.

Figure 4-1 [2]



Calculated scattering efficiency for a log normal aerosol size distribution, geometric standard deviation equal to 2, as a function of geometric mass mean radius. M is the fine particle mass concentration, b_{sp} is the scattering coefficient, and r_{gm} is the geometric mass mean radius. For reference, the right hand axis is the mass concentration required to give a visual range of 40 km.

Thus:

$$b_{ext} = b_o + (b_{ap} + b_{sp})$$

The absorption and scattering coefficients are proportional to the mass concentration of the diesel particulate, $b = AM_c$. The proportionality constant is called the extinction efficiency, referred to as A. There are several values, all in a close range, for the extinction efficiency for diesel particulate; these are listed in Table 4-1. The average value, $A = 8.0\ m^2/g$, is used for the concentration of diesel particulate. (Taking into account the carbon content of diesel particulate (approximately 70 percent), the extinction efficiency for fine elemental carbon is $11.5\ m^2/g$.) Therefore, the portion of the extinction coefficient due to diesels is the product of the increase in particulate concentration and the extinction efficiency of diesel particulate, and the equation for b_{ext} becomes:

$$b_{ext} = b_o + 8.0[m^2/g]M_c \qquad (1)$$

In order to compute percentage changes in visibility, the baseline visibility extinction, b_o, must be known. Baseline visibilities were obtained from several Trijonis reports.[6,7,8,9] Trijonis determined the existing visibilities from cumulative frequency distributions of quality-checked* airport observations. Figures 4-2 and 4-3 show the distribution of median visibilities for various parts of the country. Visibility in the Northeast tends to be rather low, with the relative humidity acting as the dominating factor. Median visibilities range from 8-12 miles with very small differences in metropolitan areas, urban/suburban areas, and nonurban areas. For the Southwest, the median visibility is 30-55 miles in large urban centers and 65-80 miles at suburban/non-urban locations. One Trijonis report [5] lists median visibility levels at 94 urban/suburban locations throughout the U.S; another report [9] lists median, tenth percentile and ninetieth percentile visibility at 12 northeastern locations. No data exists for the annual median visibility levels of all the major U.S. cities, so estimates were made from the median visibilities listed above.

The baseline visibilities (L_{vo}) are related to extinction according to the formula:

$$b_o = \frac{3.0}{L_{vo}}$$

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* Telephone surveys were conducted at the airports to ensure that each location had an adequate set of visibility markers for estimating visual range, reliable reporting practices, observation personnel and observation locations.

Table 4-1

Extinction Efficiency (A) for Diesel Particulate

Study	A (m^2/g)
Trijonis* (1982)	8.4
Klimisch (1982)	8.0
Roesslor and Faxog (1978)	6.8
Vuk, et al (1976)	8.2
Pierson (1978)	8.0

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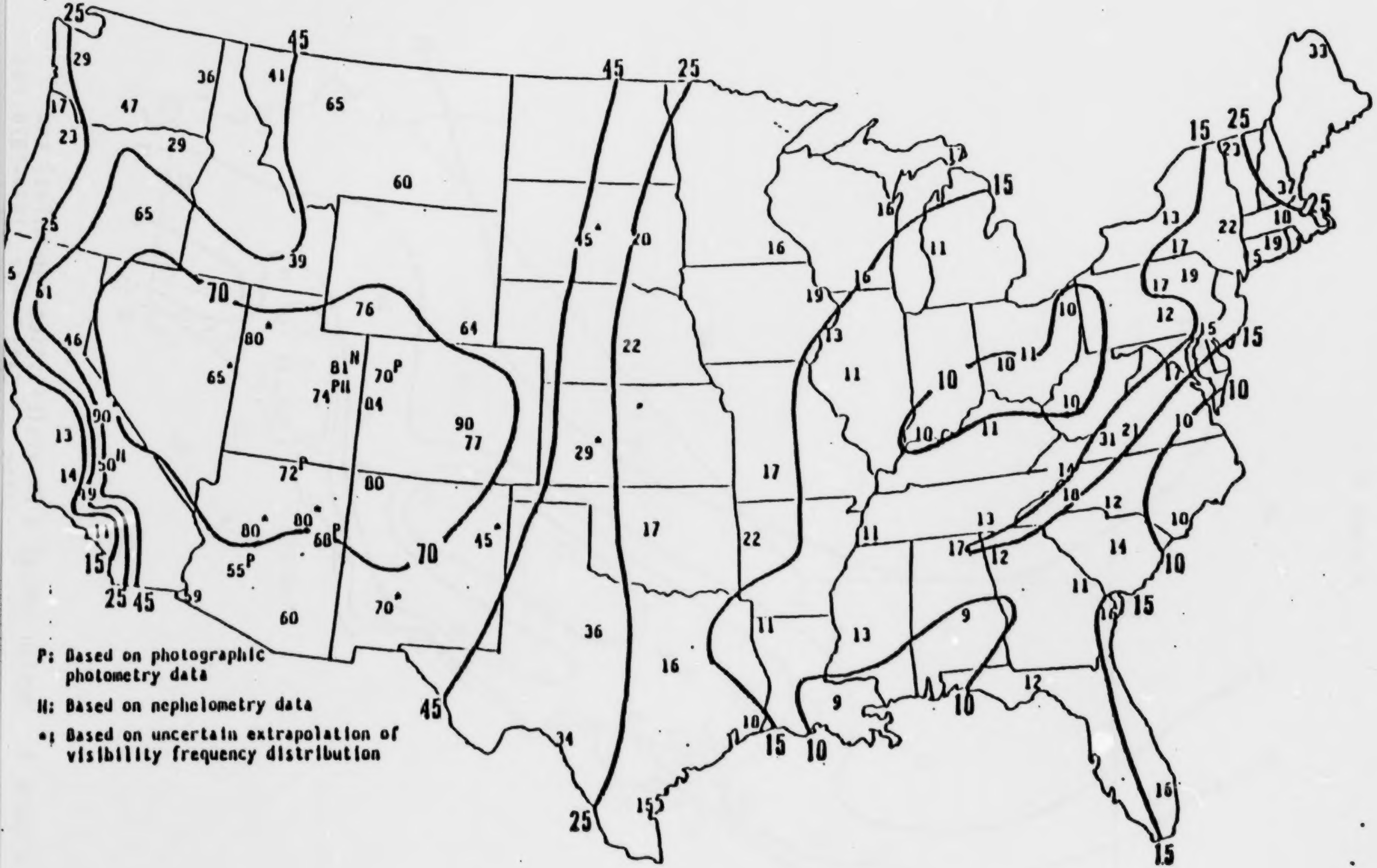
* The figure for Trijonis is calculated from his extinction efficiency for fine elemental carbon, 12 ±3 m^2/g, using a 70 percent carbon content in the diesel particulate.

Figure 4-2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P: Based on photographic photometry data
N: Based on nephelometry data
*: Based on uncertain extrapolation of visibility frequency distribution

Figure 2 Median yearly visibilities and visibility isopleths for suburban/nonurban areas..[5] 6
(miles)

Figure 4-3

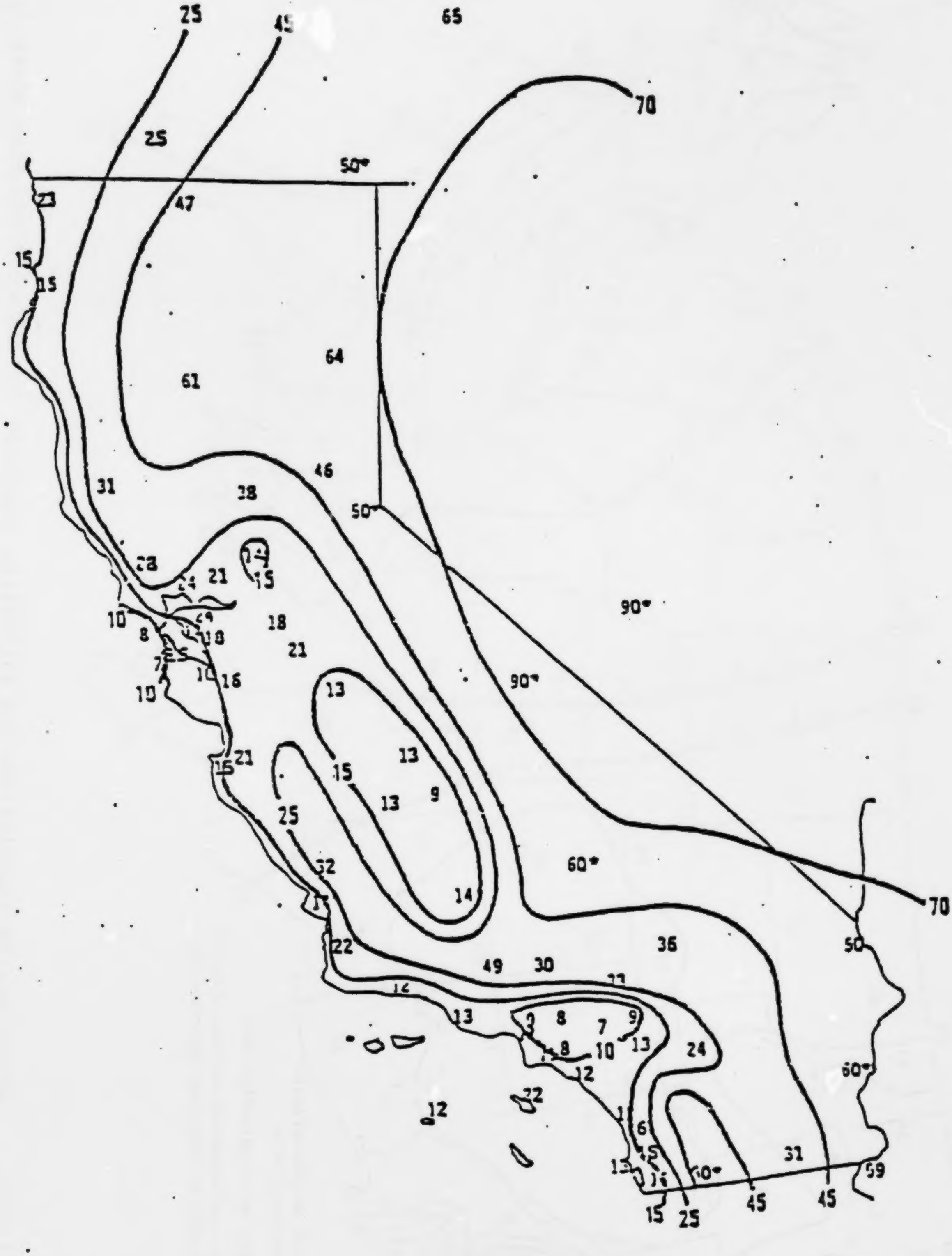
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Figure 3 Median annual 1 PM visibilities (in miles) and visibility isopleths for California, 1974-1976.[6]

The proportionality constant of 3.0 in this equation is applicable when using airport visibility data, as opposed to using the 3.91 figure from the Koschmieder formula.[6] Airport data does not adhere to the conditions for applying the Koschmieder formula because natural objects at a great distance are usually small (small objects need a contrast greater than 2 percent to be seen), and natural objects are never black. The proportionality constant of 3.0 is appropriate according to Trijonis, et al.[7]

To simplify this formula's units, it may also be expressed as:

$$b_o = \frac{18.6 \times 10^{-4}\ [\text{miles/m}]}{L_{vo}[\text{miles}]} \qquad (2)$$

where the units of extinction are inverse meters and the units of visibility are miles.

The new visual range caused by the addition or subtraction of diesel particulate can now be calculated from the following expression:

$$L_v = \frac{18.6 \times 10^{-4}[\text{miles/m}]}{b_{ext}} = \frac{18.6 \times 10^{-4}}{b_o + 8.0[m^2/g]M_c}[\text{miles/m}] \qquad (3)$$

Where:

b_{ext} is determined using Equations (1) and (2).

However, the above equation assumes that b_{ext} is constant throughout the entire visual range. This is a satisfactory assumption for the baseline situation (i.e., to estimate b_o). However, it may not be satisfactory to assume that the effect of diesel particulate will be constant throughout the visual range. The ambient concentration of diesel particulate will be relatively high in the central city and near suburban traffic corridors and will be relatively low outside of the city or metropolitan limits. Since visibility may extend to areas beyond the city or metropolitan limits, this effect must be taken into account. For those cases where visibility extends beyond the affected area, this effect may be taken into account by returning to the Beer-Lambert Law and rederiving Koschmieder's Law assuming one value of b_{ext} for a fixed distance (i.e., up to some limit) and another value of b_{ext} beyond.

IV. Revised Visibility Equations

As described earlier, the Beer-Lambert Law describes the reduction in light intensity as a function of distance and the extinction coefficient of the media. For an object outside of the affected area being viewed from within the affected area:

$$I = I_a e^{-b_a L_a}$$

Where b_a is the extinction coefficient existing within the affected area, L_a is the distance from the observer to the limit of the affected area and I_a is the light intensity of the object at the limit of the affected area. I_a is described by the equation:

$$I_a = I_o e^{-b_o (L - L_a)}$$

where L is the total distance between the object and the observer.

Combining the two equations yields:

$$I = I_o e^{(-b_a L_a - b_o(L - L_a))}$$

Applying Trijonis' application of Koschmieder's Law, $\ln(I/I_o)$ is -3.0, and solving for L (now L_v) results in:

$$L_v = \frac{3.0 - (b_a - b_0) L_a}{b_o}$$

where L and b are in inverse units, or

$$L_v = \frac{18.6 \times 10^{-4} \text{ [miles/m] } - (b_a - b_o) L_a}{b_o} \qquad (4)$$

where L_v and L_a are in miles and b_o and b_m are in inverse meters.

The term b_a can be derived using Equation (1). Substituting this into Equation (4) yields:

$$L_v = \frac{18.6 \times 10^{-4} \text{[miles/m]} - 8.0 \text{[m}^2\text{/g]} \; M_c L_a}{b_o} \qquad (5)$$

The terms b_o and M_c can be derived from Equation (2) and air quality models, respectively. Only L_a remains to be described.

L_a is the typical distance between the viewer and the limit of the affected area. Before determining this distance, the limits of the affected area must be defined. In the actual situation, the concentration of diesel particulate gradually falls off until it reaches zero; in the model being used, a constant level of diesel particulate inside the affected area and no affect outside is assumed. The limit of the affected area, L_a, must be between the point where the actual ambient concentration of diesel particulate begins to fall off and the point where it finally reaches zero. Therefore, the affected area's limit, L_a, is where the actual diesel particulate level is approximately half of its central city level. Two convenient limits which could suffice are: 1) the city limit, and 2) the metropolitan area limit. It has been assumed that the metropolitan areas and cities were circular to calculate a nominal radius.

The metropolitan area limits for large cities, such as Los Angeles (36 miles) and New York City (21 miles), appear very reasonable as diesel particulate penetration limits (i.e., for Los Angeles, L_a = 36 miles and for New York City, L_a = 21 miles). However, for smaller cities, such as Ann Arbor, Michigan (15 miles) and Madison, Wisconsin (20 miles), the metropolitan area limit appears much too large. The city limits appear much more reasonable for these smaller cities (i.e., 2.6 miles for Ann Arbor and 4 miles for Madison). Thus, metropolitan area limits will be used for the largest U.S. cities and city limits will be used for the smaller cities. This will more closely model the size of the affected areas, yielding a better model for the extent of the actual diesel particulate concentration level. To be conservative, the demarcation between the two will be made at a relatively large city population, 1,000,000, resulting in the use of metropolitan area limits in only six cases: Chicago, Detroit, Houston, Los Angeles, New York, and Philadelphia.

Now that the limits of the affected area are established, the typical distance between the viewer and the limit of the affected area must be determined. This depends on both where the viewer is located and on which direction he is viewing. While it is conceivable that a model could be formulated to determine the mean viewing distance based on relative population density and shape of the affected area, etc., the radius of the metropolitan area or city should be a sufficient estimate of the typical distance between a viewer and the limit of the metropolitan area. While most metropolitan areas are not circular, a reasonable approximation to the average distances between the center and the edge can be derived from a calculation of a nominal radius from the actual area of the metropolitan area, assuming it is circular in shape.

V. Visibility Levels

A. Methodology

Measuring the change in visibility levels due to a change in diesel particulate is dependent on four factors:

1. the mass concentration of the diesel particulate,
2. the extent of this concentration,
3. the extinction efficiency of diesel particulate, and
4. baseline visibilities.

For those cases where the visual range does not extend past the limits of the affected area, the visual range can be calculated from the following expressions:

$$L_v = \frac{18.6 \times 10^{-4}[\text{miles/m}]}{b_{ext}} \qquad (6)$$

$$b_{ext} = \frac{18.6 \times 10^{-4}[\text{miles/m}]}{L_{vo}} + 8.0[\text{m}^2/\text{g}]^* M_c$$

Where L_v is the visual range in miles, L_{vo} is the baseline visual range in miles, and M_c is the mass concentration of the diesel particulates in grams per cubic meter.

For those cases where the visual range does extend beyond the affected area, the visual range can be calculated from the following expressions:

$$L_v = \frac{18.6 \times 10^{-4}[\text{miles/m}] - 8.0[\text{m}^2/\text{g}]^* M_c L_a}{b_o} \qquad (7)$$

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* If the presence of diesel particulate is determined in terms of the elemental carbon concentration, then 11.5 m^2/g should be used instead of 8.0 m^2/g.

Where M_c is the mass concentration of diesel particulate in the affected area, L_a is the nominal radius of the affected area, and

$$b_o = \frac{18.6 \times 10^{-4} \text{ [miles/m]}}{L_{vo}}$$

where L_{vo} is the baseline visual range without the effect of diesel particulate.

B. Results

As described in the previous section, three pieces of information are needed for each city in order to project the effect of diesel particulate emissions on its visibility: 1) city radius, 2) baseline visibility, and 3) the ambient concentration of diesel particulate. The baseline visibility and the nominal radius were estimated for each city and metropolitan area with 100,000 inhabitants or more.

The ambient diesel particulate concentrations in 1995 for various cities were estimated in Chapter 3 (see Table 3-4 of that chapter). There are four concentration values relating to the four regulatory scenarios: 1) best estimate sales, relaxed controls, 2) best estimate sales, base controls, 3) worst case sales, relaxed controls, and 4) worst case sales, base controls. However, these ambient diesel particulate concentrations are not available for every city with 100,000 people or more. Thus, the available concentrations were averaged according to city size and used for those cities for which projections were not available. The four city-size categories and their corresponding average particulate concentrations are listed in Table 4-2.

Applying the baseline visibilities, nominal radii, and 1995 ambient diesel particulate concentrations to the appropriate city situation (represented by Equation 6 or 7) yields absolute visibility levels in 1995 for each of the four scenarios. From these city-specific visibility projections, the effect of 1995 diesel particulate concentrations on baseline visibility can be estimated. The average visibility reduction for each city-size category and diesel control scenario is shown in Table 4-3.

Table 4-2

Average Diesel Particulate Concentrations, $\mu g/m^3$

City Size (Population)	Best Estimate Sales		Worst Case Sales	
	Relaxed	Base	Relaxed	Base
More than 1,000,000	4.4	2.8	6.6	4.3
500,000-1,000,000	3.0	1.9	4.4	2.9
250,000-500,000	2.7	1.7	4.1	2.7
100,000-250,000	2.5	1.5	3.6	2.4

Table 4-3

Average Reduction in Visibility
Due to Diesel Particulate, Percent

City Size (Population)	Best Estimate Sales		Worst Case Sales	
	Relaxed	Base	Relaxed	Base
More than 1,000,000	20	15	27	19
500,000-1,000,000	8	6	12	8
250,000-500,000	6	4	10	6
100,000-250,000	4	2	5	3

As can be seen, the visibility impact of all scenarios is strongly dependent on city size, with the larger cities experiencing the larger effect. This is primarily due to the greater diesel particulate concentrations projected for larger cities (see Table 4-2). However, the especially large visibility effects experienced by the cities having a population of more than one million is also due to their larger estimated radii. As was described earlier, the entire metropolitan area was assumed to be affected by diesel particulate emissions in these instances, where in the cases of the three smaller groupings, only the city proper was assumed to be affected.

With respect to the various scenarios, under best estimate sales the relaxed scenario reduces visibility by 4-20 percent. These visibility reductions are reduced to 2-15 percent under the base scenario. Under worst case sales the visibility reductions under both the relaxed and base scenarios are much greater, 5-27 percent and 3-19 percent, respectively. In both cases, the base scenario removes approximately one-third of the visibility reduction of the relaxed scenario.

VI. Uncertainties in the Model

The uncertainty present in the methodology used above can bias the results in one direction or the other. Although the degree of these biases cannot be measured, it may be possible to indicate if the biases are upward or downward. This section will examine the uncertainties involved in this methodology for measuring the change in visibility caused by an increase in diesel particulate, noting the biases.

An average value of 8.0 m^2/g for the extinction efficiency of diesel particulate was drawn from several reports. This value may be a conservative estimate according to a National Research Council report.[10] As a larger extinction efficiency would yield a greater visibility impact due to diesel particulate, the currently projected impacts may be somewhat low.

The proportionality constant of 3.0 was used in place of Koschmieder's value of 3.91 to account for the real conditions present when measuring baseline visibilities as opposed to the theoretical conditions of the Koschmieder formula. Certainly this figure is an approximation and varies depending upon geographical and climatic conditions, but any resulting bias is unknown.

Diesel particulate concentrations and visibility levels were modeled as single values for each city as a simplification of the actual situation. These values assumed that all individuals in an area experienced identical levels of diesel particulate and visibility (i.e., the assumed single values).

This can lead to an overestimate of exposure for some individuals (e.g., those living inside the city area border) and an underestimate for others (e.g., those living outside the city border where the particulate level is assumed to drop off to zero). Also, visibility is strongly weather dependent and the baseline visibility levels took no account of this. However, the resulting bias is again unknown.

VII. Conclusions

A method exists to determine the visibility impact of a specific level of diesel particulate. The necessary input data include the ambient mass concentration of the diesel particulate, the extent of this concentration (assumed to be the city limit), the extinction efficiency of diesel particulate (a value of 8.0 m^3/g is used) and baseline visibilities for each city.

Visibility levels in 1995 for all U.S. cities with more that 100,000 inhabitants were projected under four regulatory scenarios. The larger cities showed a greater reduction in their visibility levels for each scenario. Under the best estimate diesel sales scenario, the relaxed control scenario resulted in a visibility loss of 4-20 percent, while the visibility reduction under the base scenario was 2-15 percent. Under the worst case diesel sales scenario, visibility decreased 5-27 percent under the relaxed scenario and 3-19 percent under the base standards. In both cases, the base scenario removed about one-third of the loss in visibility due to diesel particulate emissions under the relaxed scenario.

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CHAPTER 5

CANCER RISK ASSESSMENT

I. Introduction

Of the potential health effects associated with diesel particulate emissions, perhaps the greatest concern has been associated with its potential carcinogenic effects. This chapter will examine the state of knowledge concerning the carcinogenic potency of diesel particulate and estimate the effect of various diesel particulate control scenarios on an individual's lung cancer risk. The non-cancer health effects associated with diesel particulate are examined in Chapter 6.

The first section of this chapter reviews the major studies which have investigated the carcinogenic potency of diesel particulate. The second section compares the results of these studies and selects a likely range of carcinogenic potency for diesel particulate. The third section examines trap-oxidizer removal efficiencies of suspected carcinogens with respect to total diesel particulate matter and makes adjustments to the base scenario exposure estimates (Chapter 3). The final section combines the carcinogenic potency with the adjusted exposure values to estimate the annual lung cancer risk for an individual under each control scenario.

II. Review of Major Studies

The potential carcinogenicity of diesel particulate has been examined through both human epidemiological studies and clinical studies on animals and other lower organisms. Because the epidemiological data base is limited, much weight has had to be placed on the clinical studies. These clinical studies estimate the carcinogenic potency of diesel particulate by comparing their clinical results to the clinical results of other cancer-causing substances for which human epidemiological data are available. In this section, past and current epidemiological studies will first be reviewed, followed by a review of the comparative potency analyses.

A. Epidemiological Studies

The best means to determine the risk of developing lung cancer from a given exposure of diesel particulate is to conduct a long-term epidemiological study. Such a study would trace the health of several well-defined groups of people who were exposed to precisely known concentrations of diesel particulate for known periods of time. Comparable groups that were not exposed would also be monitored in order to detect any differences in cancer rates. The influence of such factors as diet, family history and smoking would be known in order to strengthen the validity of the study's findings and reduce the margin for error.

No matter how close the correct methodology is followed, an epidemiological study cannot "prove" the absence of a cancer hazard. Rather, a negative epidemiological study yields a statistical upper bound on the potential potency of a carcinogen under various assumptions of how the carcinogen affects cancer rates. A sound epidemiological study will only indicate that a model with an upper bound risk estimate is reasonable "in the absence of contrary information".[1] With these limitations of an epidemiological study noted, two studies of diesel particulates will be reviewed: 1) the London Transit Authority Study, which was completed a number of years ago, and 2) the U.S. Railroad Workers Study, which is still underway.

1. London Transit Authority

Of the epidemiological studies completed to date which specifically examine diesel emissions, the London Transit Authority (LTA) Study is generally considered to be the most thorough, although it too has significant deficiencies.[2] This study initially examined the lung cancer incidence among different groups of LTA employees between 1950 and 1954,[3] and was later updated to include the years through 1974.[4] Among the groups followed were diesel bus garage workers (generally high level of exposure) and design engineers (generally low level of exposure). Lung cancer incidences were identified from information on the death certificates of those who were still employed by the LTA at the time of death, ill-health retirement records, and the records of transfers to other LTA job categories. The study did not continue to monitor the health of individuals once they were no longer employed by the LTA. This is an area of potential bias since cancer typically develops several years after initial exposure to carcinogens or even after exposure terminates.

Other weaknesses of the study include the fact that the extent of individual exposure to diesel exhaust was not measured. Instead, particulate concentrations were simply measured inside and outside of selected garages on a few separate days during the 1950-74 observation period. Also, no specific cohort of employees was identified and followed throughout the study. Thus, the potential influence of such factors as smoking habits, medical history and related socioeconomic characteristics is not known.

The study found that the cancer incidence of the highly exposed group was actually less than that expected based upon Greater London lung cancer death rates in the 1950-74 timeframe. Thus, the study concluded that in regard to this study population, no evidence existed associating lung cancer to diesel engine exhaust.

However, this study has been analyzed independently by Dr. Todd Thorslund of EPA's Carcinogen Assessment Group and Dr. Jeffrey Harris, a member of the Analytical Panel of the National Academy of Sciences (NAS) Diesel Impacts Study Committee. Both found that the potential errors involved in the LTA study could have resulted in a sizeable underestimation of the carcinogenic potency of diesel particulate.[1,2]

Based on the analyses by Thorslund and Harris, it is possible that significant excess cancer deaths could result in the general population even though the LTA study showed no excess cancer deaths in the diesel particulate exposed group. Thus, the LTA study's conclusions for its population cannot be translated to the general population. Due to the potential errors involved and the lack of an upperbound estimate, this work has been disqualified from further consideration in this study.

2. U.S. Railroad Workers

Another epidemiological study is currently being conducted by Harvard University to evaluate the possible carcinogenic effect of diesel exhaust in U.S. railroad workers. Data for the study come from the U.S. Railroad Board. Components of the study include: 1) a retrospective cohort analysis of approximately 57,000 male railroad workers, 2) a case-control study of 300 incident lung cancer cases and matched controls of railroad workers, and 3) actual environmental monitoring of worker exposure to diesel exhaust. These approaches will allow for quantitative assessment of both level and duration of diesel exhaust exposure, and consideration of the major confounding factor (cigarette smoking), thus removing the major design weaknesses of the LTA study.

The retrospective cohort consists of approximately 57,000 male railroad workers, aged 40-64 in 1959, with 10-20 years of railroad service at that time. These workers were selected from job categories having high diesel exposure and an appropriate sample of control exposure categories. The massive amount of data being generated in the retrospective and case-control studies is currently being analyzed. Air pollutants being monitored in five round-house locations include nitrogen dioxide, sulfur dioxide, carbon monoxide, respirable and non-respirable particulate and its constituents, such as sulfates, polycyclic aromatic compounds and other organic compounds. In addition, fractions of the particulate sample extracts will be analyzed by mutagen bioassays, such as the Ames test. A qualitative comparison of automobile diesel exhaust with the railroad diesel exhaust will be performed by correlating the gas chromatography/mass spectra of the polycyclic aromatic compounds of each.

A pilot study was undertaken to evaluate the feasibility of this larger study. The cohort of the pilot study consisted of approximately 2,500 male railroad workers who were between the ages of 45 and 64, working in 1967 and who had at least 10 years of railroad service. Of these workers, 69.8 percent were in occupations exposed to high concentrations of diesel exhaust. The risk ratio for lung cancer in diesel-exposed workers relative to unexposed workers was 1.42, or a 42 percent increase. However, this increased risk was not statistically significant due primarily to the small size of the cohort.[5] The larger retrospective cohort study and the case-control study are currently in progress and are scheduled for completion in the near future; at the time of this writing, no formal results have been published.

B. Comparative Potency Analyses

Due to the limited epidemiological data available, estimations of the human lung cancer risk from diesel particulate have been made using a comparative potency method developed by EPA.[6] In this comparative potency method, the results of non-human laboratory bioassays are used to compare the carcinogenic and mutagenic potencies of diesel particulate (specifically, the particle-bound organics) with those of other combustion and pyrolysis products that have been shown by epidemiological data to cause lung cancer in humans. Estimates of the human lung cancer risks from exposure to these established carcinogens, based on epidemiological studies, can then be adjusted by the corresponding estimates of their potencies relative to diesel particulate to yield estimates of the lung cancer risk from diesel particulate. The equation used is given below.

$$\begin{array}{l}\text{Estimated}\\ \text{Human Risk}\\ \text{(diesel)}\end{array} = \begin{array}{c}\text{Human Risk}\\ \text{(carcinogen)}\end{array} \times \frac{\text{Bioassay Potency (diesel)}}{\text{Bioassay Potency (carcinogen)}}$$

The ratio of potencies obtained from the same bioassay is referred to as the relative potency. The unit risk exposure, referred to in terms of constant exposure per year, always assumes a lifetime exposure. The unit risk is calculated for each emissions source in order to have a basis for comparing carcinogenic potency.

This methodology is based on the assumption that the ratio of the carcinogenic potency of diesel emissions to the potencies of proven carcinogenic products is preserved across human and non-human biological systems. Although this assumption has not been proven and is a novel approach to risk assessment, EPA has determined that the comparative potency approach is the most promising method available until a reliable epidemiological study focusing on exposure to diesel exhaust is performed.

The human carcinogens (comparative sources) selected by EPA were coke oven emissions, roofing tar emissions and cigarette smoke condensate. The mobile source samples selected included those from a HDDE (Caterpillar 3304), three LDDVs (Datsun Nissan 220C, Oldsmobile 350, and Volkswagen turbocharged Rabbit), and a gasoline-fueled, catalyst-equipped vehicle (Ford Mustang II). Data from other LDDVs were also reviewed and will be summarized later in this section. The organics extracted from the particulate emitted from the above-mentioned sources were used to determine the relative potencies.

The comparative sources and the mobile source samples were both tested in mutagenesis and carcinogenesis bioassays. The mutagenesis bioassays selected included reverse mutation in Salmonella typhimurium (Ames test), forward mutation in L5178Y mouse lymphoma cells, forward mutation in Balb/c 3T3 mouse embryo fibroblasts, forward mutation in Chinese hamster ovary cells, mitotic recombination in Saccharomyces cerevisiae, DNA breakage in Syrian hamster embryo cells, and sister chromatid exchange in Chinese hamster ovary cells. The carcinogenesis bioassays included oncogenic transformation in Balb/c 3T3 cells, viral enhancement of transformation in Syrian hamster embryo cells, and skin initiation and skin carcinogenicity in SENCAR mice. Further details of the study design can be found elsewhere.[7]

The potencies obtained in these bioassays, together with epidemiological data on the comparative sources, were combined to estimate the human lung cancer risk from diesel particulate in three independent analyses performed by Dr. Jeffrey Harris, Lovelace Biomedical and Environmental Research, and EPA. Each will be discussed below. The analyses differ with respect to the choice of bioassays selected for determination of the relative potencies and the choice of comparative source epidemiological data.

It should be noted that EPA did not conduct new epidemiological studies as part of this approach, but rather relied upon existing data. For coke ovens, the work of

Mazumdar[8] and Land[9] was used, for roofing tar emissions Hammond's[10] data were applied, and that of Dell and Peto[11] were used in the case of cigarette smoke. The Harris and Lovelace analyses relied upon the same coke oven and roofing tar data. For cigarette smoke, Lovelace used the data of USHEW,[12] Hammond[13] and Kahn,[14] which resulted in a risk estimate similar to that obtained by EPA. Harris did not include cigarette smoke as a comparative source in his analysis.

1. Harris

In addition to his analysis of the London Transit Authority Study, Harris conducted a comparative potency analysis for the National Academy of Sciences.[1] The comparative source emissions selected by Harris were coke oven emissions and roofing tar emissions. Using a linear relative model, Harris analyzed the epidemiological data for coke oven and roofing tar emissions to obtain estimates of the proportional increase in lung cancer incidence per unit of cumulative lifetime exposure to coke oven emissions (0.044) and roofing tar emissions (0.015).

Harris used data from three short-term bioassays to estimate the relative potencies of the diesel (light-duty only) and comparative source samples. The bioassays used were tumor initiation in SENCAR mice by skin painting, enhancement of viral transformation in Syrian hamster embryo cells, and mutagenesis in L5178Y mouse lymphoma cells. The results from these bioassays can be found in Tables A-1 and A-2 of the Appendix. Harris then applied these relative potencies to his estimates of the proportional increase in lung cancer incidence from exposure to coke oven and roofing tar emissions to obtain estimates of the proportional increase in lung cancer incidence from exposure to diesel emissions.

Harris' overall estimate was a 0.0035 percent proportional increase in lung cancer incidence per unit exposure (i.e., one microgram per cubic meter of diesel particulate for one year assuming lifetime exposure). This is roughly equivalent to 1.4 x 10^{-6} incidences of lung cancer per person per year due to a continuous lifetime exposure of one microgram per cubic meter of diesel particulate.*

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* The proportional increases in lung cancer incidence obtained by Harris were translated into absolute measures of lung cancer incidence independently by Thorslund.[6]

2. Lovelace Biomedical and Environmental Research

Lovelace used two methodologies to estimate the cancer risk from exposure to LDD particulate.[15]

The first method assumed that diesel particulate was not more mutagenic or carcinogenic than the most potent of coke oven, roofing tar or cigarette particulate. First, the annual lung cancer risk per person for each of the three carcinogens was estimated from the epidemiological studies of coke oven workers, roofers, smokers and nonsmokers using a linear, nonthreshold model. Then, the average concentration of each type of particulate inhaled over a year was estimated and used to estimate the annual unit lung cancer risk per individual for these comparative sources. All of these figures are presented in Table 5-1. Lovelace then assigned a figure of 1.5×10^{-6} lung cancers per person due to a lifetime exposure of one microgram per cubic meter of diesel particulate as an upper estimate of the potency of diesel particulate. This figure was based primarily on the estimated annual unit risks for coke oven and roofing tar particulate, which are both between 1.0×10^{-6} and 1.5×10^{-6}.

The second method used the bioassay data developed by EPA to estimate the relative potencies of the LDD and comparative source samples. Like Harris, these relative potencies were then applied to the unit risks derived from the epidemiological studies of the known carcinogens.

The comparative sources selected from the EPA work were coke oven emissions, roofing tar emissions and cigarette smoke condensate. Urban soot was also selected independently by Lovelace as an additional comparative source. The mutagenesis bioassays used were the Ames assay, forward mutation in Chinese hamster ovary cells (HGPRT gene locus assay), forward mutation in L5178Y mouse lymphoma cells, and forward mutation in Balb/c 3T3 mouse embryo fibroblasts. The carcinogenesis bioassays used were oncogenic transformation in Balb/c 3T3 cells, viral enhancement of transformation in Syrian hamster embryo cells, and skin initiation and skin carcinogenicity in SENCAR mice. These bioassay data are presented in Table A-3 of the Appendix.

The overall relative potencies resulting from a comparison of the data in Table A-3 are shown in Table 5-2, along with the annual unit risks already presented in Table 5-1 and the estimated annual unit risks for diesel particulate resulting from each comparison. When only the comparative sources used by EPA are considered (coke oven, roofing tar and cigarette smoke condensate), the annual unit risk estimates for diesel particulate range from 0.07×10^{-6} to 0.6×10^{-6} lung

Table 5-1

Summary of Inhalation Exposures and Annual Lung Cancer Risks for Surrogate Populations - Lovelace*

Study Population	Average Air[a] Concentration of Particles (mg/m^3)	Annual Lung Cancer Risk x 10^6 (per person, per year)	Annual Risk x 10^6 (per person, per ug/m, per year)
Coke Oven Workers	3	4000	1.3
Roofers	1	1100	1.1
Smokers: (cigarettes/ day)			
1-9	2-16	260	0.03
10-19	18-35	470	0.02
20-39	36-71	800	0.02
40+	73+	1070	0.01
Urban Nonsmokers	0.06	70	1.2
Rural Nonsmokers	0.03	30	1.0

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* Information in this table was excerpted from Reference 15.

[a] The average air concentration of particles was estimated as the total mass of particles inhaled per year divided by all of the air breathed per year (assumed to be 20 m^3 per day X 365 days per year).

Table 5-2

Lung Cancer Risk From Exposure to Diesel Exhaust Based Upon Relative Potencies of Surrogate Substances - Lovelace*

Surrogate Exposure	Median Relative Potency (surrogate/ diesel ratio)	Annual Cancer Risk X10^6 (per person per ug/m^3)	Estimated Risk of Diesel Particlate X10^6 (per person per ug/m^3)
Coke Oven Emissions	5	1.3	0.3
Roofing Tar Vapor	2	1.1	0.6
Cigarette Smoke Condensate	0.3	0.02	0.07
Urban Soot	0.4	1.2	3.0
Selected Diesel Lung Cancer Risk			1.0

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* Information on this table was excerpted from Reference 15.

cancers per person per year due to a constant lifetime exposure of one microgram per cubic meter of diesel particulate (unit exposure). When urban soot is also considered as a comparative source, the range increases to 0.07 x 10^{-6} to 3.0 x 10^{-6}.

Based on the results of both methods, Lovelace chose 1.0 x 10^{-6} as being the most representative estimate for the annual unit lung cancer risk due to diesel particulate.

3. Environmental Protection Agency (EPA)

Members of EPA's Office of Research and Development also recently estimated the annual unit cancer risk of diesel particulate using a comparative potency method very similar to that used by both Harris and Lovelace.[16] The comparative sources used in this analysis were coke oven, roofing tar and cigarette smoke. Epidemiological data for coke oven workers, roofing tar workers and cigarette smokers were examined using a linear, nonthreshold model to determine the annual unit lung cancer risk for each carcinogen. A summary of these risk estimates can be found in Table A-4 of the Appendix.

The relative potencies of the coke oven, roofing tar, cigarette smoke condensate and mobile source samples were evaluated by a large number of bioassays which have already been described. The bioassays used in the final determination of the relative potencies were the tumorigenicity bioassays involving skin initiation and skin carcinogenicity in SENCAR mice, the Ames bioassay, the L5178Y mouse lymphoma cell mutagenesis bioassay, and the sister chromatid exchange (SCE) bioassay in Chinese hamster ovary cells. The results from these tests are given in Tables A-5 and A-6 of the Appendix. It should be noted that the mobile source and comparative source samples were also evaluated in a number of additional bioassays. The bioassays used in this analysis (and those selected in the Harris and Lovelace studies) were selected for their ability to produce dose-related effects and the strength and relevance of the end point being measured. The relative potencies are shown in Tables A-7 and A-8.

Two steps were subsequently followed to determine the lung cancer risks for the diesels and the gasoline vehicle. First, the relative potencies in the mouse skin tumor initiation assay (Table A-7) were used to obtain the annual unit risk estimate for the Nissan particulate from the annual unit risks for the coke oven, roofing tar and cigarette smoke condensates (Table A-4). Second, the annual unit risks for the other diesel particulates were obtained by multiplying the annual unit risk of the Nissan particulate by the net relative diesel potencies of Table A-8, which were based on the Ames, lymphoma and SCE bioassays.

The annual unit lung cancer risk estimates resulting from these calculations are shown in Table 5-3. Since the organics extracted from the particulate were used in the bioassays, the risk estimates were calculated in terms of organics and then converted in terms of particulate. For the three LDDVs, the annual risk estimates per person range from 0.26 x 10^{-6} to 0.46 x 10^{-6} due to lifetime exposure to one microgram per cubic meter of particulate. The LDDV with the highest risk estimate, the Nissan, was initially selected as representative of abnormally high particulate organic mutagenic activity for an upper range value. Although engines of this type are reported to have faulty injector systems that result in the high activity emissions, other studies show a number of LDDVs with Ames mutagenicity activities above that reported for the Nissan.[15] Hence the Nissan risk estimate is not unreasonably high for a diesel vehicle.

It is interesting to note that emissions from the Caterpillar HDDE had about one-tenth the potency of the LDDVs. This may be due to two reasons: 1) the particulate from the Caterpillar had been stored for more than a year before use, and 2) except for the Caterpillar, all the vehicles were operated on a chassis dynamometer using the highway fuel economy test cycle (HWFE7). The Caterpillar engine was tested on an engine dynamometer and operated at low load conditions (Mode II using a steady-state operation of 2200 rpm and an 85-pound load), possibly resulting in low activity. Due to these non-representative conditions, the risk estimate for the Caterpillar particulate will not be used further.

Ames test results on diesel particulate samples other than those cited in the above analysis were also reviewed. Major emphasis was on light-duty diesel vehicles, particularly in-use automobiles, to ensure the representativeness of the above-described results. Due to availability of data, and because the TA-98 strain was used in the earlier analysis, data for this particular strain were of primary interest.

Results of four separate studies on the specific mutagenic activity of diesel particulate are summarized in Table A-9 of the Appendix[17-20]; numbers of revertants per ug of SOF (soluble organic fraction) are given for samples both with and without metabolic activation. Means for the specific activities range from 0.97 to 13.7 for the non-activated samples, and between 2.08 and 8.66 revertants/ug SOF for the metabolically-activated tests.

A comparison of these supplementary data with the values used to determine cancer risk estimates earlier in this section shows fairly strong agreement. The specific activities listed

Table 5-3

Unit Lung Cancer Risk Estimates for Diesel Particulate - EPA*

Diesel Source	Unit Risk Estimates (annual risk/ug/m^3)	
	Organics	Particulate
Nissan[a]	0.58×10^{-5}	0.46×10^{-6}
Volkswagen Rabbit[b]	0.17×10^{-5}	0.30×10^{-6}
Oldsmobile[b]	0.16×10^{-5}	0.26×10^{-6}
Caterpillar[b]	0.87×10^{-7}	0.024×10^{-6}

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* This table was excerpted from Reference 16 in which lifetime risks were presented. These risks have been converted to annual risks by dividing by the median lifespan (76.2 years).

[a] Based on average relative mouse skin tumor initiation activity (Table A-7).

[b] Based on average relative activity in the mouse lymphoma, SCE, and Ames Bioassays (+MA) (Table A-8).

for diesels in the comparative risk study (Table A-5) average 5.7 and 5.8 revertants/ug SOF for the non-activated and activated strains, respectively, excluding the Caterpillar data; this compares to overall means of 5.3 and 6.1 in Table A-9. A comparison of maximum specific activities shows 11 and 13 revertants/ug for non-activated and activated samples, respectively, in Table A-5, versus 21 and 13 in Table A-9 (again excluding the Caterpillar). These maximums support the judgment that the Nissan estimates from Table A-5 are not unusually high and are reasonable estimates of maximum potencies for diesel vehicles. The close agreement between the overall means in both tables add further support for the representativeness of the data used in calculating the cancer risk estimates for diesel particulate and for exclusion of the Caterpillar data.

III. Choosing a Value or Range of Values

A summary of the risk estimates obtained from the three comparative risk studies is shown in Table 5-4. It should be obvious from the preceding discussion and a comparison of the figures in Table 5-4 that there is no concensus among the scientific community as to the carcinogenic potency of diesel particulate. The three potency studies differ at nearly all possible points: 1) the estimated annual unit cancer risks of the known carcinogens, even though, for the most part, the same epidemiological data are used, 2) the non-human bioassays selected for actual derivation of the relative potencies, and 3) the relative weightings given to those bioassays selected.

In addition, all of these studies rely on the assumption that the relative carcinogenic potencies of diesel emissions and the related environmental emissions are preserved across human and non-human biological systems. Although this assumption has not been proven correct, it is the best one that can be made until a reliable epidemiological study focusing on exposure to diesel exhaust is performed.

It should also be noted that all of the comparative potency analyses discussed used a linear, nonthreshold dose-response model to extrapolate cancer incidence to lower doses. While this has been the most widely used model in the past, others are gaining more use presently. Figure 5-1 depicts two typical examples of other models: an infralinear model and a linear, threshold model.[21] Since all the exposures simulated in the non-human laboratory tests are very high to demonstrate effects with small number of specimens, the results must be extrapolated downward to lower, more realistic doses. Examining Figure 5-1, the point common to all three models can be taken to be the result of the high-dose

Table 5-4

Summary of Lung Cancer Risk Estimates

Comparative Potency Analysis	Annual Risk x 10^6 (per person per ug/m^3 particulate)
Harris	1.4
Lovelace	1.0[a]
EPA	0.26-0.46[b]

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[a] When the EPA comparative sources were used, the risk estimates obtained by Lovelace range from 0.07 x 10^{-6} to 0.6 x 10^{-6}. When urban soot was also considered by Lovelace as a comparative source, the risk estimates range from 0.07 x 10^{-6} to 3.0 x 10^{-6}. Lovelace chose 1.0 x 10^{-6} as being most representative.

[b] Since the heavy-duty Caterpillar sample is not considered representative, the range of risk estimates is restricted to the range of risk estimates obtained for the light-duty vehicles.

bioassay. Then, as can be seen, both the infralinear and linear, threshold model result in lower, low-dose risks than the linear, nonthreshold model. Because of this, depending on which model is correct, the use of the linear, nonthreshold model could overestimate the cancer risk at lower doses. To date, however, the linear, nonthreshold model has been the only one applied to diesel particulate emissions.*

Because of the lack of consensus among the various studies, this study will use the range of risk estimates obtained from the comparative potency analyses of Harris, Lovelace and EPA. Referring to Table 5-4, the range of risk estimates selected for this analysis is 0.26×10^{-6} to 1.4×10^{-6} lung cancers per person per year due to a constant lifetime exposure of one microgram per cubic meter of diesel particulate.

IV. Reduction in Cancer Risk Via Total Particulate Control

In Chapter 3, individual annual exposures to diesel particulate matter were estimated for the relaxed and base scenarios (Table 3-8). The reduction in exposure between the relaxed and base scenarios reflects projected trap-oxidizer efficiencies in removing total diesel particulate (namely those of non-catalyzed traps). In the interest of projecting cancer risk from exposure to diesel particulate emissions, the actual removal of suspected carcinogens should be compared to the overall reduction in total particulate mass.

This section will concentrate on the effects a particulate trap has on the soluble organic fraction (SOF) of diesel particulate. It is this SOF, particularly the heavier hydrocarbons, that is usually associated with the bulk of bio-activity; this bio-activity, in turn, is generally considered an indication of carcinogenic tendencies. Part A of this section will evaluate the trap's ability to control SOF with respect to total particulate. Part B considers possible changes in mutagenic or carcinogenic tendencies of the organics with the use of a trap. Part C will evaluate data from previous sections in order to determine a ratio of removal efficiencies (SOF to total particulate matter); in turn, this ratio will be used to adjust the exposure figures from Chapter 3.

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* There is one additional model, the supralinear model, which actually results in a higher, low-dose risk than the linear, nonthreshold model;[21] however, its application to diesel particulate would be the furthest from being established of all of the other models.

The data examined in this section were limited to studies performed with non-catalyzed, ceramic particulate traps. This distinction was made in an effort to examine perhaps the "worst" case with respect to carcinogen removal. Non-catalyzed traps primarily filter solid particulate matter; heavy hydrocarbons can still be in the gaseous phase at the trap and can escape and later either adsorb onto the remaining particulate or remain in the gas phase. Catalyzed traps reduce gaseous hydrocarbons preferentially to particulate matter. Therefore, the use of catalysts in particulate traps would be expected to reduce cancer risk to an even greater extent than they reduce total particulate matter. As the costs of non-catalyzed traps are used in Chapter 8, non-catalyzed traps were chosen here to represent the reduction in cancer risk due to base scenario control measures.

At this time, it is important to note that data examining various carcinogenic parameters both with and without trap use were very limited in their availability. This study bases its preliminary conclusions on existing data only; no new tests were initiated. Some areas of the study (such as soluble organics removal) had more pertinent data available than did others; therefore, in drawing conclusions, more weighting was given to these areas. New information may become available in the future that could modify these results.

A. SOF Removal Efficiencies

For all the available data examined, it was found that actual emission rates for soluble organics were reduced substantially with the particulate trap in place; however, the SOF removal efficiency was, in most cases, somewhat lower than that for total particulate reduction. Due to this lower SOF efficiency, the "after-trap" SOF as a percent of total mass was generally found to be higher than the "before-trap" SOF percentage. For HDDVs, average reduction efficiencies for total particulate and SOF were 76 and 68 percent, respectively; for LDDVs, the total efficiency of 80 percent was somewhat higher than the SOF removal of 69 percent. For both classes, the efficiency "gap" was approximately the same (8 to 11 percent). (See Tables A-10 and A-11 in the Appendix for test data on SOF emission rates.[22-26])

When the actual fraction of soluble organics (expressed as a percent of total mass) is examined before and after the trap, the "with-trap" SOF tends to be higher than the baseline percentage. The increase in percent SOF for the LDDVs is substantially higher than that shown with the HDDVs; the SOF as a percent of total particulate increased (on the average) by 23.4 percent and 11.4 percent for light and heavy vehicles, respectively (see Tables A-12 and A-13).[22-26]

There is one concern related to simply focusing on the SOF. The SOF is made up primarily of unburned organics that condense or adsorb onto the solid particulate matter; organics that do not attach to solids remain in the gaseous phase. Since the trap removes the great majority of the particulate to which unburned HCs can attach, one concern is that more organics will remain in the gaseous phase. This concern was addressed by examining available data on gaseous HC emissions with and without a trap. In six LD and three HD studies, 90 percent of the data showed either a decrease in HC emissions or no change at all with the use of a trap; in all instances except one, any increases were less than three percent higher than baseline data (see Tables A-14 and A-15). Overall, gaseous HC emissions were reduced by 12.4 percent and 22.2 percent (LD and HD averages, respectively). Furthermore, a durability test on a LD vehicle [23] showed that HC removal efficiencies did not deteriorate with increased VMT; HC reductions ranged from 12 to 54 percent, with an average of 33.5 percent (see Table A-16). The overall indication here is that particulate traps do not force organics back into the gaseous phase; this, coupled with the data showing substantial decreases in SOF emissions, leads to a preliminary conclusion that the number of cancer-causing organics being released to the atmosphere will most likely be significantly reduced with the use of a trap.

In summary, SOF removal efficiencies were substantial in all cases, although generally lower than overall particulate removal efficiencies. Related to this, data indicates that soluble organic material as a percentage of total particulate was slightly higher after the trap than before. Further examination of the SOF will consider changes in bio-activity, and relatedly, cancer-risk.

B. Mutagenic Activity

There are various methods of assessing the cancer risks associated with the SOF of diesel particulate matter. One of the most widely-used methods is the Ames test, which measures bio-activity in terms of revertants per mass of SOF tested; when these values are converted to units of revertants per mile or per kilowatt-hour, they are generally considered meaningful estimates of the carcinogenic properties of the SOF. A second method of estimating cancer risk is to examine the temperature distribution of the SOF; the focus, with respect to carcinogenicity, is placed on the heavier hydrocarbons (three or more benzene rings), which require higher temperatures to vaporize. A final indication of cancer-risk is the measure of benzo-a-pyrene (BaP) emissions; this poly-nuclear aromatic hydrocarbon is a known human carcinogen. However, it is well known that BaP represents only a minor fraction of the potential carcinogenicity of diesel particulate; therefore, a change in the concentration of this one compound may not indicate a similar change in overall carcinogenic potency.

Ames mutagenicity test data both with and without a non-catalyzed trap were available for one light-duty diesel vehicle and one heavy-duty diesel engine.[27,25] In general, the number of revertants per mile or per kilowatt-hour (LDDV and HDDE, respectively) was lower with the trap in place; the average reduction for LDDVs was 7.5 percent, while the value for HDDEs was 39.1 percent. The reductions in total particulate mass were 72 and 89 percent, respectively. Thus, the trap removed revertants 10 and 44 percent as efficiently as it removed total particulate mass.

The actual number of revertants per mass of SOF was found to increase with the use of a trap; light-duty bio-activity increased by an average of 113 percent, while the heavier engine had an increase of 170 percent. (See Tables A-17 and A-18 for the Ames test data on LDDV and HDDE, respectively.)[27,25]

Boiling-point temperature distributions were also available for a light-duty automobile and a heavy-duty engine. In both cases, data supported the finding that the particulate trap was most able to capture the heavier HCs, presumably including the mutagenic, multi-ring hydrocarbons as well. In other words, the "after-trap" SOF seemed to contain fewer heavy HCs than the "before-trap" SOF samples.

Actual temperature distribution data were not available for the above-mentioned LDDV; however, curves for "with" and "without" trap were provided which plotted volatile particulate mass (SOF) versus TGA temperature.[24] The "with-trap" curve levels off at approximately 300°C, while the "without-trap" curve continues to rise until a temperature of 500°C is reached. Therefore, the "without-trap" sample contained more materials with high boiling points than did the "with-trap" SOF.

Temperature data was available for a heavy-duty diesel coach engine and is given in Table A-19.[26] This boiling point distribution of the two SOF samples shows that the "with-trap", SOF has approximately the same fraction of lower-boiling compounds as the "without-trap" sample (discounting the initial value), but a lesser amount of high-boiling compounds. Again, the data indicate that the trap was able to remove some of the carcinogenic, heavy HCs being emitted as part of the SOF.

One final indicator of carcinogenic tendencies, the amount of benzo-a-pyrene (BaP), will be examined for a heavy-duty diesel engine and an in-service coach. Data showing the change in BaP level due to trap use are found in Table A-20.[26] As shown, engine tests showed emissions of BaP to increase by 0-250 percent, while vehicle tests showed BaP emissions to increase 10-175 percent with the trap. (It should be noted that BaP is difficult to measure, and the measurements were only quoted to one or two significant figures.)

C. Exposure to Potential Carcinogens

Diesel particulate exposure levels for the base scenario estimated in Table 3-8 take into account only reductions (from the relaxed levels) of total particulate mass. In order to project cancer risk for the base scenario, the exposure figures must be adjusted to take into account the reduction in suspected carcinogens relative to overall particulate removal. Table A-21 summarizes trap removal-efficiencies for the available indicators of carcinogenic potential: SOF, mutagens, and BaP.

The data show a wide variation between the reduction in the three indicators of carcinogenic potential and that of total particulate. For several reasons, the data on soluble organic emissions were considered the most pertinent. Seven independent studies were available and the results were very consistent. Overall, the average ratios of SOF removal relative to total particulate removal ranged between 0.7 and 1.0. The temperature distribution data indicate that the trap removes the heaviest organics preferentially. Since the bulk of bio-activity is believed to be associated with these heavier organics, the reduction in SOF may actually underestimate the reduction in bio-activity.

The little Ames test data that compared emissions with and without trap use showed inconsistent results. The most data was provided by a HD study on a Caterpillar engine [25]; Table A-18 summarizes the results. The figures calculated for the last four modes are consistent with earlier SOF reductions; excluding the first two modes, the ratio of reduction in mutagenic activity with respect to overall particulate removal ranges from 0.79-1.0. Ames test data on LD vehicles was very limited and not very consistent (Table A-17); the average reduction of 7.5 percent was not representative of previous findings, and, therefore, this data was not weighted very heavily in calculating the adjusted exposure levels.

There appear to be a number of reasons not to place much weight on the BaP data. The only BaP emissions data available were from one HD study of a coach engine and coach vehicle. The engine in both cases was a DDA 6V-71N, which has unique emissions characteristics and has already been replaced by the 6V-92TA or 8V-92TA in transit buses. Also BaP only accounts for a very small fraction of the overall bio-activity. Given these factors plus the inconsistency between the BaP results and all the other data, very little weighting should be given to this data in determining new exposure levels.

In summary, the reductions in SOF emissions were considered the most representative, and, therefore, the highest weighting was given to results from these tests. In view of added support from the Ames test results on heavy-duty vehicles and the temperature distribution data, it appears that the reductions in SOF emissions should be indicative of a reduced cancer-risk. However, since removal efficiencies for the SOF are 70-100 percent of that for total particulate, a factor of 0.7 to 1.0 will be used to adjust the base scenario exposure levels from Table 3-8. The adjusted exposure levels are shown in Table 5-5.

V. Estimated Risk Based on Projected Diesel Exposure

The range of potency estimates for diesel particulate derived in Section III can be combined with the adjusted scenario-specific particulate exposures from Section IV to yield estimates of the individual lung cancer risk due to diesel particulate in 1995. After this has been done, these individual lung cancer risk estimates will be compared to cancer and accidental risks from other sources.

A. Scenario-Specific Individual Lung Cancer Risks

The population exposures to diesel particulate from both light- and heavy-duty vehicles in 1995 were derived in Chapter 3 for four scenarios: 1) best estimate diesel sales with the relaxed control scenario, 2) best estimate diesel sales with the base control scenario, 3) worst case diesel sales with the relaxed control scenario, and 4) worst case diesel sales with the base control scenario. Subsequently, in this chapter, the exposures for the base scenarios were adjusted to reflect reductions in carcinogens with respect to total particulate removal.

The potency estimates of Table 5-4, based on the linear nonthreshold extrapolation model, only require the annual average individual exposure to obtain estimates of annual average cancer risk per individual. The projected nationwide

Table 5-5

Individual Diesel Cancer Risk Projections in 1995

	Scenario			
	Best Estimate Sales		Worst Case Sales	
	Relaxed	Base	Relaxed	Base
Projected Individual Diesel Particulate Exposure in 1995 (ug/m^3)				
Light-Duty	1.8	1.5	3.7	2.9
Heavy-Duty	3.2	1.6	3.7	1.9
TOTAL	5.0	3.1	7.4	4.8
Adjusted Exposures Based on Ratio of Carcinogen vs. Total Particulate Removals with Particulate Trap (base scenario)				
Light-Duty	1.8	1.5-1.6	3.7	2.9-3.1
Heavy-Duty	3.2	1.6-2.1	7.4	1.9-2.4
TOTAL	5.0	3.1-3.7	7.4	4.8-5.5
Estimated Individual Risk Based on Adjusted Diesel Particulate Exposures in 1995 X 10^6 (lung cancer risk/person-year)*				
Light-Duty	0.5-2.5	0.4-2.2	1.0-5.2	0.8-4.3
Heavy-Duty	0.8-4.5	0.4-2.9	1.0-5.2	0.5-3.4
TOTAL	1.3-7.0	0.8-5.1	1.9-10.4	1.3-7.7

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* Individual lung cancer risks in 1995 were obtained by multiplying the adjusted individual diesel particulate exposure in 1995 for each scenario by the range of potency estimates for diesel particulate (0.26×10^{-6} - 1.4×10^{-6} risk/person-year-ug/m^3).

annual average exposure levels for individuals living in urban areas in 1995 for each scenario, expressed in terms of micrograms per cubic meter, are found in Table 5-5. These exposure estimates are then simply multiplied by the range of individual potencies, expressed as lung cancer risk per micrograms per cubic meter per year, to obtain the range of estimated individual lung cancer risk in 1995 due to diesel particulate exposure under each scenario.

The resultant individual lung cancer risks in 1995 for each scenario are also shown in Table 5-5. Individual lung cancer risks in 1995 due to exposure to particulate from both light- and heavy-duty diesels range from 0.8×10^{-6} to 7.7×10^{-6} under the base control scenarios and 1.3×10^{-6} to 10.4×10^{-6} under the relaxed control scenarios.

As can also be seen from Table 5-5, the relative contribution of LDD emissions is much greater assuming worst case diesel sales than best estimate sales. Also, the individual lung cancer risk is reduced by roughly 27-38 percent under the base scenario relative to the relaxed scenario. The effect of the base scenario is greatest with respect to the HDD contribution.

B. Comparison of Diesel Cancer Risk with Other Risks

To place these estimated cancer risks in perspective, they can be compared to current (generally 1981) individual risks from other sources. The other individual risks provided for comparison include commonplace (accidental) risks of death[28-30], most of which would be considered involuntary (unavoidable), and cancer risks from exposure to various sources.[30-32] Also included is the risk of death from lung cancer for smokers whose deaths are attributable to smoking, along with the risk from lung cancer for the general population whose deaths are attributable to causes other than smoking.[33] These risks, expressed as individual cancer risk or probability of death per year, are given in Table 5-6.

Accidental risks are generally applicable to the entire U.S. population. As can be seen in Table 5-6, the aggregate risk for tornadoes, floods, lightning, tropical cyclones and hurricanes is within the same order of magnitude as that given for diesel particulate. In contrast, the risks of not wearing seat belts, burns, drowning and motor vehicle accidents all exceed the risk projected for exposure to diesel particulate. The risk of a motor vehicle accident is more than an order of magnitude greater than the maximum risk estimated for diesel particulate.

Table 5-6

Comparison of Risks from Various Sources*

Sources of Risk	Estimated Risk (risk/person-year)	Exposed Population
Diesel Particulate:		Urban U.S.
Best Estimate Sales:		
Relaxed Scenario	1.3×10^{-6} - 7.0×10^{-6}	
Base Scenario	0.8×10^{-6} - 5.1×10^{-6}	
Worst Case Sales:		
Relaxed Scenario	1.9×10^{-6} -10.4×10^{-6}	
Base Scenario	1.3×10^{-6} - 7.7×10^{-6}	
Commonplace Risks		
Motor Vehicle Accident[28]	222.0×10^{-6}	Entire U.S.
Not Wearing Seat Belts[29]	112.0×10^{-6}	Entire U.S.
Drowning[28]	26.0×10^{-6}	General U.S.
Burns[28]	21.0×10^{-6}	Entire U.S.
Tornados, Floods, Lightning, Tropical Cyclones and Hurricanes[30]	2.0×10^{-6}	General U.S.
Cancer Risks		
Natural Background Radiation (sea level)[30]	20.0×10^{-6}	Entire U.S.
Average Diagnostic Medical X-Rays in the United States[30]	20.0×10^{-6}	Widespread
Frequent Airline Passenger (4 hours per week flying)[30]	10.0×10^{-6}	Limited
Four Tablespoons Peanut Butter Per Day (due to presence of aflatoxin)[30]	8.0×10^{-6}	Fairly Widespread
Ethylene Dibromide[31]	4.2×10^{-6}	Widespread
One 12-Ounce Diet Drink Per Day[30]	2.6×10^{-6}	Widespread
Arsenic[32]	1.7×10^{-6}	1% of U.S.
Miami or New Orleans Drinking Water (due to presence of chloroform)[30]	1.0×10^{-6}	Southern U.S., Urban
Lung Cancers:		Entire U.S.
For Smokers Due to Smoking[33]	419.0×10^{-6}	
For General Population Due to Causes Other Than Smoking[33]	73.9×10^{-6}	

In addition to the accidental risks discussed above, cancer risks which result from dietary and occupational exposures are included for comparison. These cancer risks are roughly within the same order of magnitude as that for diesel particulate. (The risk from lung cancer will be discussed separately.) Exposures to many of the cancer risks given in Table 5-6, including the risk from diesel particulate, can be applied across the general U.S. urban population or a vast majority of it. Exposures to the other cancer risks such as arsenic, or frequent airline travel, can only be applied to a selected segment of the population. For example, only 2.82 million people, or roughly 1 percent of the population are exposed by virtue of their occupation to atmospheric arsenic.[32] Thus, the number of people exposed to arsenic is far less than those exposed to diesel particulate and the other cancer risks whose exposures can be applied across the general U.S. population. The number of people exposed to each source should be taken into consideration when making direct comparisons of risk.

In some cases, risks resulting from certain occupational exposures far exceed those risks presented in Table 5-6. For example, exposures to arsenic results in an individual annual risk of respiratory cancer as high as 180 x 10^{-6} for those few workers exposed near cotton gins.[32] For ethylene dibromide, cancers can result from both dietary and occupational exposures. The risk from dietary exposures to ethylene dibromide is given in Table 5-6. The occupational risks of cancer resulting from inhalation of ethylene dibromide vapor can be as high as 5.2 x 10^{-3} for citrus warehouse laborers.[31]

The risk of lung cancer for smokers whose deaths are attributable to smoking, along with the risk from lung cancer for the general population whose deaths are attributable to causes other than smoking, are also included in Table 5-6 for comparison. The maximum lung cancer risk given for diesel particulate is roughly 2.5 percent of the lung cancer risk for smokers whose deaths are attributable to smoking, and 14 percent of the lung cancer risk for the general population whose deaths are attributable to causes other than smoking. The analogous figures for the minimum diesel exposure are 0.2 percent and 1.1 percent, respectively. As can be seen, smoking is the primary contributor to lung cancer deaths in the U.S. (85 percent).

CHAPTER 6

NON-CANCER HEALTH EFFECTS OF DIESEL PARTICULATE

I. Introduction

One of the primary concerns regarding diesel particulate is its potential for adversely affecting human health. The potential adverse health effects of this material can be divided into two broad categories: 1) carcinogenic and 2) non-carcinogenic, or non-cancer. This chapter deals specifically with non-cancer health effects. The potential carcinogenic effects of diesel particulate were already discussed in Chapter 5.

Although a large amount of information documenting the adverse health effects of inhaling particulate matter is available in the literature, comparatively little deals specifically with diesel particulate. However, concern over the potentially adverse health effects of exposure to diesel exhaust has recently increased, and has resulted in a significant amount of new research concerning diesel particulate and its effects on health.[1,2] Unfortunately, because much of the diesel particulate health effects information which is available is comparatively recent and has not been peer reviewed by other scientists, very few conclusive statements can be made regarding the health effects of diesel particulate exposure.[3] Therefore, at this time, the best approach for evaluating the non-cancer health effects of diesel particulate is to evaluate the health effects of particles for which established literature is available. However, before outlining how this comparative analysis will be performed, it is important to describe three things this analysis will not do.

First, the fact that particles in the ambient air can cause adverse non-cancer health effects will not be established here. It has long been recognized that exposure to various forms of particulate matter can cause a wide variety of adverse non-cancer health effects. These effects have been well documented in the literature and total suspended particulate matter was among the first airborne pollutants to have a NAAQS established by EPA in 1971.

Second, in evaluating the documented non-cancer health effects of particulate matter, the focus will not be on any specific types of particulate, but rather on typical ambient mixtures of particles. Obviously, some types of particulate affect health differently than others. For example, soluble particles may affect health through different mechanisms than insoluble particles. Some specific particles also are inherently more dangerous than others (e.g., radioactive material). However, because it is generally impossible epidemiologically to ascribe the adverse health effects of ambient exposures to any specific component of the particulate mixture, the effects of specific particles are less important than the effects of typical mixtures of particles found in the atmosphere.

Third, this comparative analysis will not be conducted quantitatively, but qualitatively. While a few quantitative health effects studies based on measurements of total suspended particulate or British smoke shade are available, the extrapolation of these results to diesel particulate could only be based on qualitative relationships and the quantitative results would imply a degree of precision beyond that which was defendable.

Proceeding to the description of what will be done, the comparative analysis will be performed on two levels: particulate inhalation characteristics and laboratory health effects testing. The available information on each of these levels will be presented first for ambient inhalable particulate and second for diesel particulate, with a comparison of the two sets of results following on each level. An overall asessment will then be made as to whether or not diesel particulate should be expected to affect health (non-carcinogenically) disproportionate to its impact on ambient mass particulate levels.

II. Non-Cancer Health Effects of Typical Particulate Matter

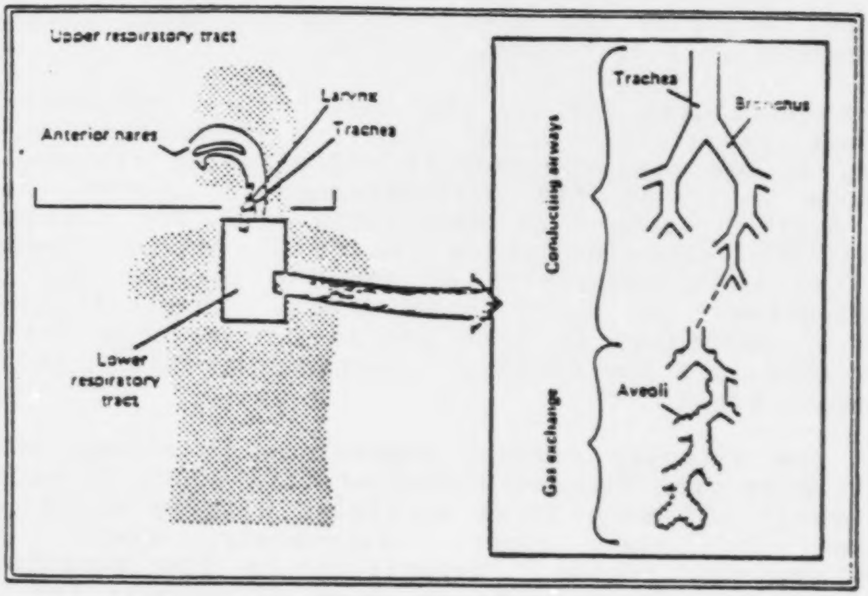
A. Inhalation of Particulate Matter

One of the most easily understood determinants of adverse health effects from inhaling particulate matter is the "body dose." For the purposes of this chapter, the important aspects of body dose are: 1) where particles are deposited in the respiratory tract, and 2) how these particles are cleared from the system by natural defense mechanisms. Therefore, some general knowledge regarding the structure of the respiratory tract, in addition to deposition and clearance within the system is a prerequisite for specifically discussing the non-cancer health effects of particulate exposure.

The principal features of the respiratory system are depicted in Figure 6-1. The upper respiratory tract begins with the nares (or mouth during oral breathing) and ends at the entrance to the trachea. The lower respiratory tract is subdivided into the conducting airways (or tracheobronchial region) and the gas-exchange region (or alveolar region). The tracheobronchial region consists of the trachea down to the minute terminal bronchioles. The alveolar region includes the partially alveolated bronchioles and finally terminates with the alveoli themselves. (A more complete description of the respiratory tract as it relates to particle deposition can be found in Reference 5.)

Figure 6-1

Diagrammatic Representation of the
Human Upper and Lower Respiratory Tract[4]


Upper respiratory tract
Larynx
Anterior nares
Trachea
Lower respiratory tract
Conducting airways
Trachea
Bronchus
Gas exchange
Aveoli

1. Deposition in the Respiratory Tract

As stated above, the health effects associated with particulate matter in the respiratory tract are dependent, to a large degree, upon where in the tract deposition takes place. Spatial deposition within the respiratory system is primarily determined by particle size, with the mode of breathing (nose versus mouth) also having a substantial effect on the disposition of large particles.

Moving through the respiratory tract, deposition in the upper respiratory tract during nose breathing is nearly 100 percent complete for particles with diameters larger than about 10 micrometers and declines to about 10 percent for particles with diameters less than 1 micrometer.[5] During mouth breathing, deposition in the upper respiratory tract is less efficient, although the vast majority of large particles are still removed in this region.

Most particles smaller than about 10-15 micrometers enter the lower respiratory tract and are deposited, to varying degrees, in the tracheobronchial and alveolar regions as shown in Figure 6-2. In the tracheobronchial region, deposition during mouth breathing is especially high for particles with diameters of 5-10 micrometers (up to 80 percent removal) and tapers off to a deposition of about 5 percent for particles with diameters of 0.1-1 micrometers. Deposition of 5-10 micrometer particles in this region during nose breathing is considerably less due to their previous deposition in the upper respiratory tract.

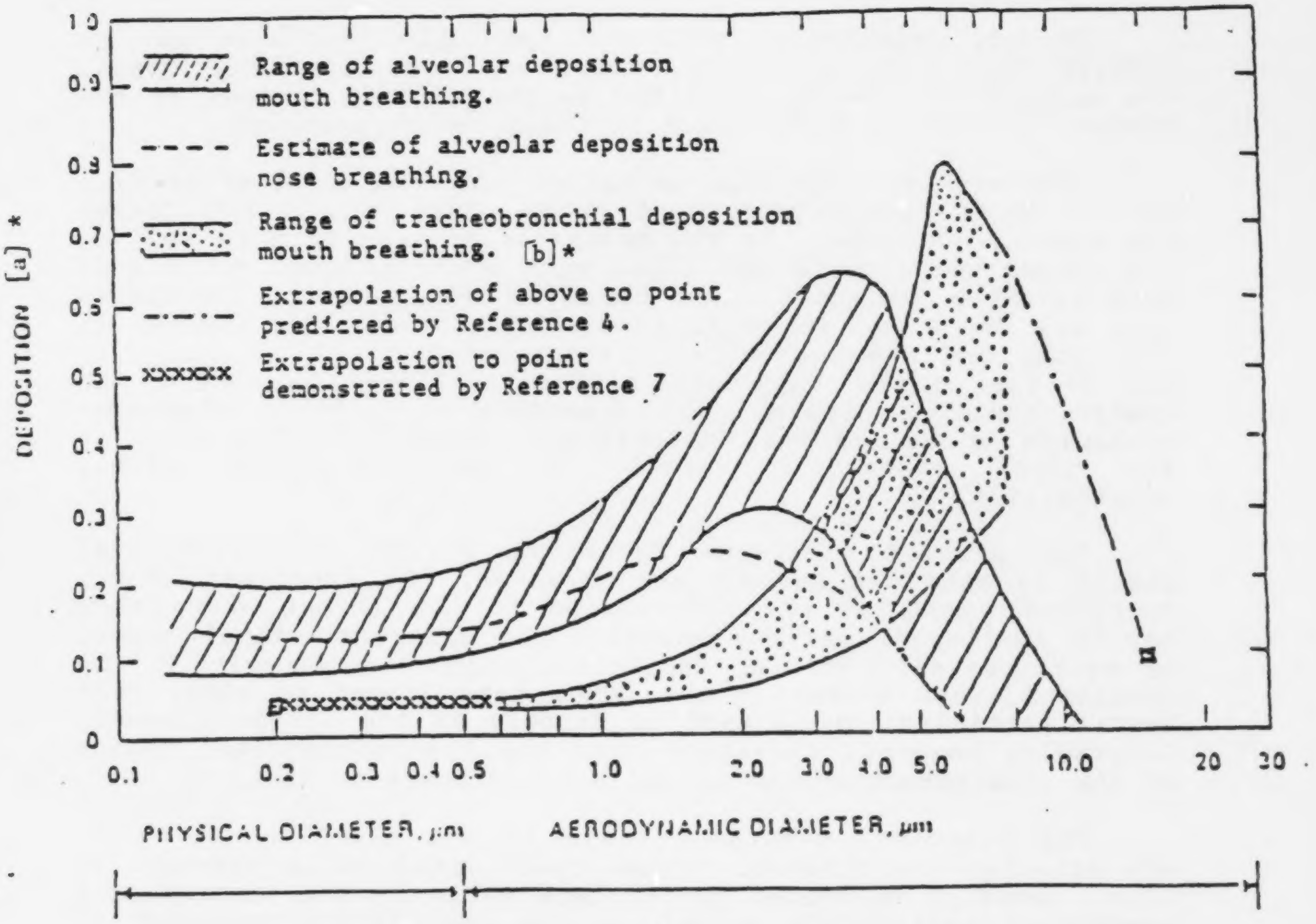
In the alveolar region, deposition is almost nonexistent for particles with diameters greater than about 10 micrometers, since nearly all such large particles already would have been deposited in the upper respiratory tract and the tracheobronchial region. Deposition in the alveolar region during mouth breathing peaks at about 65 percent for particles with diameters of 3-4 micrometers and declines to around 15 percent for particles with diameters between 0.1-0.2 micrometers. This peak is still present during nose breathing, but it's level is much less (25 percent). Generally, this information shows that particles with diameters less than about 10-15 micrometers generally penetrate deeper into the respiratory system than larger particles.

2. Clearance of Particulate Matter From the Respiratory Tract

Clearance is the process whereby particles are removed from the respiratory tract. This process is described in this section in a simplified manner. It must be noted, however, that the mechanisms for removing particles are often complex and the efficiencies of these mechanisms often vary significantly between individuals, due to such factors as

Figure 6-2[6,7]

Deposition in the Tracheobronchial and Alveolar Regions By Indicated Particle Diameter



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[a] Deposition is expressed as fraction of particles of a given diameter entering the mouth (or nose).

[b] **Tracheobronchial** deposition during nose breathing likely would be less than that depicted for mouth breathing.

smoking, pathological abnormalities, and response to inhaled pollutants. A more complete description of respiratory clearance is available in Reference 2.

Particulates may be removed from the respiratory tract in two principal ways. First, particles which are soluble in body fluids (or the soluble coating on insoluble particles) may dissolve in any region of the respiratory system where deposition occurs. After dissolution, the constituents of the particle may interact locally with cells or tissues, or they may be absorbed into the blood and transported to other areas of the body.

Second, relatively inert and insoluble particles may be removed from the respiratory tract by more mechanical means. The process is somewhat specific to the various regions of the system; therefore, each region is discussed separately.

Clearance of insoluble particles from the anterior portion of the upper respiratory tract takes place mainly by blowing the nose or sneezing. In the posterior portion of this region, the conducting airways are lined with both ciliated cells that have hairlike projections and mucus-secreting cells. Particles that are deposited in these conducting airways are trapped in the mucus and are mechanically transported by cilia action to the throat, where they are either swallowed, entering the gastrointestinal tract, or expectorated. This clearance mechanism is called the "mucociliary conveyor." Clearance in the upper respiratory tract is normally rapid (i.e., minutes).[6]

The primary clearance mechanism in the tracheobroncial region is also the mucociliary conveyor. As described above, entrained particles are transported to the throat where they may be swallowed, thereby entering the gastrointestinal tract, or expectorated. Smaller particles, which may deposit in the smaller airways deeper in the lung, take longer to clear than larger particles, which tend to deposit in the larger airways. Generally, however, clearance from the tracheobronchial region of the respiratory system normally takes hours to days.[6]

The principal clearance route in the alveolar region is via alveolar macrophages. These specialized cells phagocytize (i.e., engulf) deposited particulate matter. Some macrophages containing particulate travel to the mucociliary conveyor of the tracheobronchial region where they are cleared through the gastrointestinal tract. Others travel to lymph nodes and are cleared from the body through the lymphatic system. Clearance of insoluble particles from the alveolar region generally takes months or years.[6]

3. Related Health Concerns

There are two principal concerns associated with the deposition of inhalable particulate in the lower respiratory tract. First, particles deposited in this area, even if not directly toxic themselves (e.g., inert particles), may have hazardous materials adsorbed onto their surfaces. Consequently, these adsorbed, hazardous materials may be transported deep into the most sensitive areas of the lung where they may cause localized effects or be absorbed and circulated to other parts of the body, causing problems elsewhere. Second, all particles deposited in this area have relatively long residence times. As discussed previously, clearance of particles in the tracheobronchial region may take days, while in the alveolar region it may take years to clear insoluble particles. These long residence times provide a greater opportunity to generate health problems even if toxic materials are not present. Both of these concerns have led EPA's Office of Air Quality Planning and Standards to recommend that a NAAQS be established for particulate matter with diameters of 10 micrometers or less.[6] Therefore, the particles in the ambient air which are associated with the effects of concern have two general characteristics: 1) chemical constituents that are soluble in body fluids, and 2) diameters of 10 micrometers or less.

These general characteristics of typical particulate matter that cause adverse non-cancer health effects are important later in this analysis, since the greater the similarity between this particulate matter and diesel particulate, the stronger the inference that diesel particles can also cause adverse health effects. The key points to remember are:

1. Deposition in the respiratory system is particle-size dependent,

2. Smaller particles with diameters less than about 10-15 micrometers are transported into the deepest portions of the respiratory system (tracheobronchial or alveolar regions) where they reside for long periods of time (hours to years), and

3. Within this subset of inhalable particles, some particles are deposited in greater amounts depending on their diameter and the heights and breadths of these peaks are dependent on the mode of breathing (mouth versus nose).

B. Effects of Particulate Deposition in the Lower Respiratory Tract

As stated in the previous section, the deposition of inhalable particulate in the tracheobronchial and alveolar regions of the respiratory tract pose the greatest threat to health. The effects of concern in the tracheobronchial region

1. Reduced lung function,

2. Aggravation of existing respiratory disease (especially for bronchitics and asthmatics),

3. Increased infectious disease, and

4. Predisposition to the development of bronchitis.[6]

In the alveolar region the effects of concern include:

1. Reduced lung function,

2. Damage to lung tissues,

3. Increased susceptibility to infection, and

4. Aggravation or predisposition to cardiopulmonary diseases.[6]

These effects have been observed to varying degrees in laboratory and epidemiological studies. Because of individual variation and limitations in analytical methodologies, it is difficult to tell at what particulate concentrations these effects begin or become significant. Presently, many of these effects do not appear to have clear thresholds.[6]

The exact causes of many of the above non-cancer health effects are not well known, but the following mechanisms or responses are generally involved either singly or in combination:[5,6]

1. Macrophage damage due to physical overloading with particles or because of a toxic response to chemicals adsorbed on particles;

2. Excess mucus secretion causing a reduction in the flow rate of the mucociliary conveyor;

3. Structural changes in the lung tissue due to physically or chemically induced damage;

4. Deposition of particles in excess of the lung's clearance ability with an attendant build-up of particles; and

5. Bronchioconstriction of airways due to the stimulation of nerves in the tracheobronchial region.

While the health effects listed above are a step closer to overall human health than the lung functions (mechanisms) just described, it is the list of mechanisms which will be most useful below in assessing the relative potency of diesel particulate. There are simply not enough data on the effect of diesel particulates on the types of health effects listed above. While the amount of available data on the effect of diesel particulate on lung function is also less than desireable, it is greater than that on health effects and will provide the basis for comparison below.

III. Non-Cancer Health Effects of Diesel Particulate Matter

A. Inhalation of Diesel Particulate

Concerns regarding the health effects of ambient exposures to diesel particulate were first based on its physical and chemical characteristics. The particulate matter from diesel engines is composed of basic units which are 0.1 micrometer or less in diameter.[8] These units form agglomerates with diameters ranging up to a maximum of about 1 micrometer. Most of the agglomerates, however, are significantly smaller than 1 micrometer in diameter (90 percent by mass), with about 50 percent by mass being 0.3 micrometer or less.[8,9,10] The small size of diesel particulate means that it is deposited in the lower respiratory tract, where clearance may take years.

Also important is the fact that the basic particulate unit is composed of a carbonaceous core with a wide variety of organic compounds adsorbed onto its surface. While at least one study specifically identified 70 organic compounds associated with diesel particulate,[8] the great majority of the individual compounds remains unknown. Such chemical constituents could react locally with the cells or tissues of the lung, or be transported to other areas of the body.

These are the same general characteristics that were identified above for typical inhalable particles. Therefore, based solely on the inhalation characteristics of diesel particulate, it is logical to expect that exposure to diesel particulate could cause the same adverse non-cancer health effects as other inhalable particulate.

B. Effect of Diesel Particulate Deposition in the Lower Respiratory Tract

This inhalation-based connection between inhalable particulate and diesel particulate has fostered research specifically aimed at understanding the non-cancer health effects of exposure to diesel particles. The results of this research can be used to resolve two issues which are of paramount concern. First, does diesel particulate actually elicit the same adverse effects or responses that were described above for inhalable particulate in general, as would be expected based on the similarities between the particles? Second, is exposure to diesel particulate disproportionately more hazardous than would be suggested by its contribution to the concentration of inhalable particulate suspended in the ambient atmosphere because of its deep lung deposition and adsorbed chemicals? More specifically, is the potency of diesel particulate and the mixture of inhalable particulate in the ambient air significantly different, so that any increase in diesel particulate beyond current levels would be especially hazardous? These two questions are discussed separately because one issue can be resolved more conclusively than the other at this time; the first question in this section and the second in the next.

Two types of studies which specifically deal with diesel emissions are most useful in answering either of these questions: epidemiological and laboratory. Before the findings of these studies are presented, it should be noted that most of this research has already been compiled or reviewed in References 3, 11, and 12. Because of this, only a brief overview of the literature will be presented here.

The epidemiological research into the non-cancer health effects of diesel particulate exposure is extremely limited. There are no studies which specifically evaluate diesel particulate. Only a very few studies evaluate diesel exhaust, and diesel particulate by association. The primary reason for this is the lack of suitable populations available for study.[1]

Some of the studies that have been completed suggest that occupational exposure to diesel exhaust (e.g., railroad, transit, mining workers) results in a higher prevalence of chronic respiratory symptoms, bronchitis, and loss of lung function.[3] Other studies have shown no significant adverse effects between groups of exposed and unexposed individuals.[3] Therefore, although the available epidemiological studies suggest that chronic exposure to diesel exhaust, including diesel particulate, may adversely affect health, the results are inconclusive. Because of this, no firm conclusion regarding the health effects of diesel particulate can be made based on this type of information. Thus, the results of laboratory studies must be examined to better determine the effects of diesel particulate exposure.

Most laboratory investigations of diesel particulate exposure have been conducted at higher particulate concentrations than normally would be encountered in the natural environment. This is common practice in such studies and is done to reduce the cost of such research. Because of the high exposures used in these studies, they are very useful in identifying the mechanisms or responses that would account for the effects of concern that are observed in the "real world" (e.g., bronchitis and infectious disease). However, they are less useful for identifying health effects that will occur at realistic exposure levels.

Most of the laboratory studies involving diesel particulate have shown, to varying degrees, the same basic effects on lung function that were previously described for inhalable particulate matter, including alveolar macrophage damage, excess secretion of mucus, lung tissue damage, possible adverse effects on the immune system, and particle build-up in the lung.[3,11,12] This similarity of response provides strong evidence that exposure to diesel particulate has the potential to elicit many of the same adverse health effects which were also previously described for inhalable particulate in general. Therefore, the original concerns regarding diesel particulate that were based simply on its inhalation characteristics are supported by more recent direct evidence.

C. The Hazard of Diesel Particulate Relative to General Inhalable Particulate

The issue of diesel particulate's relative hazard is a more difficult issue to resolve. As discussed above, the few quantitative epidemiological studies are not useful to characterize the non-cancer health effects of diesel particulates because their results are inconclusive. Also, the use of very high particulate concentrations in the laboratory studies generally precludes using this research to evaluate the health risk of ambient exposures to diesel particles in comparison to that associated with the ambient mixture of particles. Nevertheless, some studies have investigated the systemic toxicology of diesel exhaust. Such studies are particularly useful in evaluating the concern that the organic chemicals adsorbed on the surface of diesel particulate may make it disproportionately more hazardous than other particulate in the ambient mixture.

Generally, the results of these studies have not demonstrated any significant gross toxicological effects from exposure to diesel particulate.[8] A possible explanation for this lack of effect is that other research has suggested that although the organic layer of diesel particulate is soluble in body fluids, it may be released very slowly and that enzyme systems in the lungs may metabolize these chemical consitituents into more innocuous substances.[1] Therefore, this information suggests that the organic layer of diesel particulate may not cause significant non-carcinogenic toxicological effects. Until more information is generated, however, the possibility that diesel particulate may be disproportionately more hazardous cannot be dismissed.

Information concerning the efficiency with which particles of various sizes are deposited in the lower respiratory tract may also provide some insight into the relative hazard of diesel particulate. It was previously stated that almost all diesel particulate is smaller than 1 micrometer in diameter.

Figure 6-2 shows, for example, that the deposition for these sized particles in the alveolar region during mouth breathing is substantially less than for particles with diameters of 1-6 micrometers. (The effect is present, though less dramatic, for nose breathing.) Therefore, particles in the ambient air which are somewhat larger than diesel particulate may pose a slightly greater health hazard on the basis of mass deposited in the lower respiratory tract. This suggests that, on a mass concentration basis alone, diesel particulate may not be more hazardous than would be accounted for by its contribution to the total ambient mixture of inhalable particulates, and that it could be slightly less hazardous than certain larger, though still inhalable, particulate. Here again, however, the information is simply too limited to make any conclusive judgments.

IV. Summary and Conclusions

Based on the available health effects and deposition studies, there is no direct evidence that diesel particulate is disproportionately more potent in causing non-cancer health effects than an equivalent mass of the current ambient mixture of particles. However, this information is so limited that it does not provide a sufficient basis for conclusively eliminating the concern that diesel particulate may be more hazardous because of its chemical composition and deep lung deposition. Therefore, the issue of diesel particulate's relative hazard cannot be fully resolved at this time. Ongoing research may shed more light on this issue in the future.

The following overall conclusions regarding the non-cancer health effects of diesel particulate are possible, based on the information summarized above.

1. Laboratory studies have shown diesel particulate matter has the potential to cause or contribute to adverse health effects such as reduced lung function, damage to lung tissues, increased suceptibility to infection, aggravation of existing respiratory disease, predisposition to bronchitis, and aggravation of or predisposition to cardiopulmonary disease.

2. There is insufficient evidence to conclusively judge whether diesel particulate is or is not more hazardous than the mixture of various particles suspended in the ambient air with diameters of 10 micrometers or less (i.e., inhalable particulate). The very limited information from health effects and deposition studies suggests that diesel particulate may not be more hazardous under certain conditions (i.e., mouth breathing). However, until more data becomes available, diesel particulate should be considered as harmful in causing non-cancer health effects as the ambient mixture of inhalable particles.

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CHAPTER 7

SOILING EFFECTS

I. Introduction

With the increased use of diesel-powered vehicles, the impact of diesel particulate emissions on materials has become a subject for investigation. The major material effect associated with chemically non-reactive atmospheric particles, such as diesel particulate, is that of material soiling.[4] This chapter will examine the effects of diesel particulate on soiling.

In the past, the vast majority of soiling studies have dealt with general atmospheric particulate, while little work has been done specifically on the soiling impact of diesel particulate. However, by considering the relative characteristics of diesel particulate, it is possible to adapt the findings of studies addressing atmospheric particulate soiling to diesel particulate soiling.

The soiling effect caused by increased ambient levels of diesel particulate can be addressed in a number of ways. One approach would be to derive three relationships: 1) a relationship between ambient particulate levels and the physical phenomena of soiling (i.e., particle deposition), 2) a relationship between soiling and cleaning frequency, and 3) a relationship between cleaning frequency and cleaning costs. By combining the three, a relationship between ambient particulate levels and the cost associated with removing the soiling can be obtained. However, with this approach intermediate relationships are also determinable (i.e., the relationship between particulate levels and cleaning frequency). A second approach would be to derive a single relationship between ambient particulate levels and the cost of soiling. This latter methodology usually utilizes surveys of individuals' intentions or actions to determine a "willingness-to-pay" associated with a decrease in soiling. Deriving a relationship between ambient particulate levels and the behavior of the people affected by soiling is still another approach; this approach may utilize property values in its methodology to determine the cost associated with a reaction to soiling.

This analysis will not address any economic costs associated with soiling due to the controversy connected with the existing economic soiling analyses. Instead, this analysis will only address the practical aspects of soiling (i.e., soiling as a function of particulate concentration and cleaning (or other soiling remedy) frequency as a function of soiling).

This restriction in scope has the unfortunate side effect of placing the great majority of the research addressing atmospheric particulate soiling outside the scope of this study. The remaining research primarily addresses the effect of total suspended particulate (TSP) on soiling, with little having been done on the effect of soiling on cleaning frequency or on the soiling effects of various subclasses of TSP. No experimental research has been conducted on soiling by diesel particulate.

Given this, this study will take a three-step approach to address the issue of diesel particulate soiling. First, the physical process of soiling will be defined and described. Second, studies addressing soiling by TSP will be reviewed to assess the current state of knowledge in the area. Third, soiling by diesel particulate will be compared to that by TSP by comparing the physical and chemical properties of both types of particulate and postulating their effect on soiling. The goal of the entire process will be to arrive at some relative value for the soiling effect of ambient diesel particulate to that of TSP.

II. Description of the Soiling Process

Soiling is defined as the build-up of a layer of deposited atmospheric particulates on an exposed surface.[1] A soiled surface appears dirty to the eye and, as the layer of deposited particulates increases, it will become detectable by touch. Characteristics associated with soiling are: 1) a loss of reflectance of visual light by an opaque material surface, or 2) a reduction in light transmission through a transparent material.

The time interval required to transform horizontal and vertical surfaces from a clean to a perceptibly dirty state is generally determined by particle composition and the rate of deposition. This process is also influenced by the location and spatial alignment of the material, the texture and color of the surface relative to the particle, and meteorological variables like moisture, temperature and wind speed.[2]

The degree of soiling is determined by measuring reflectance from an opaque surface and by measuring haze through a transparent surface (window glass is the most common transparent surface). The greater the original reflectance of the surface, the more observable the soiling will be.[3] This can easily be seen by imagining the effects of soiling on a white-painted surface, which has a reflectance of more than 90 percent, as compared to the effects of soiling on a black-painted surface, with a much lower reflectance. Of course the soiling effects on a dark surface by a white or light-colored particle is equally observable, but because diesel particulate is dark, this is not a concern.

III. Atmospheric Particulate Soiling

A small number of studies have been performed relating TSP levels to the physical rate of soiling. This section will briefly review four such studies. The first two studies were experimental in nature and simply attempted to determine the relationship between particulate concentration, time, and soiling. The third study used surveys and attempted to go one step further by relating particle concentration to the frequency of soiled removal (in this case, painting). The fourth study, a literature review, identified those cleaning tasks that would be affected by increased soiling resulting from increased ambient particulate levels.

In the first study, Parker attempted to determine the relationship between changes in the reflectance of a surface and the accumulation of particles.[1] Reflectance changes of white painted surfaces showed a first order dependence upon total pollutant dosage as defined by the expression:

$$R = R_p + (R_o - R_p) \exp(-KCt)$$

Where:

R = reflectance of the surface,
R_o = initial reflectance of the surface,
R_p = reflectance of particles,
K = deposition rate constant,
C = particle concentration,
t = exposure time.

It is interesting to note that, if soiling is defined as the change in surface reflectance ($R_o - R$) rather than simply the surface reflectance (R), the above equation becomes:

$$R_o - R = (R_o - R_p)(1 - \exp(-KCt))$$

This is the equation for exponential decay, which, among other processes, describes the decay of radioactive materials. The change in reflectance is rapid at first and slows as time goes on. The final reflectance of the surface approaches the reflectance of the particulate assymptotically (i.e., very gradually). A doubling of the particle concentration would not affect the final reflectance of the surface, but would double the rate of soiling. This is shown in Figure 7-1.

In a second, similar study, Beloin and Haynie exposed six materials to particulate soiling.[4] A linear regression analysis resulted in the following relationships for two of the materials:

1. For acrylic paint:

$$R_o - R = 92.5 - R = 1.36C^{.345}t^{.612}$$

2. For white asphalt shingles:

$$R_o - R = 41.8 - R = .0078C^{1.007}t^{.595}$$

where the units of C and t are micrograms per cubic meter and months, respectively.

Here, soiling $(R - R_o)$ is dependent on certain powers of both particle concentration and time. While these relationships appear quite different from that put forth by Harker, they are not entirely inconsistent. First, Beloin and Haynie were actually addressing a situation quite different from that addressed by Harker. Beloin and Haynie's experiments and their correlations included a variety of particulate types, all having different properties. Harken's relation only applies to a single type of particulate. Second, the powers associated with particle concentration and time in the Beloin and Haynie equations are all essentially between zero and one, which is what would be expected if the process described by Harker was examined for a specific period of time. The fact that the powers for concentration and time are not equal is more of a question, as Harker's model implies they should be the same. However, the fact that Beloin and Haynie included different types of particulate in their study could be the explanation.

To illustrate this possibility, a portion of the data from the Beloin and Haynie study and their equation for acrylic paint have been reproduced in Figure 2. A specific instance of Harker's equation was then fit to the data. As can be seen, the two relations agree very well and both describe the data adequately. Thus, while the exact form of the relationship between soiling and particle concentration is not known, it is clear that atmospheric particulate does result in soiling and that an increase in particle concentration will increase the degree of soiling, and very likely to the same degree (i.e., a doubling of particulate will double the soiling).

A relationship between the frequency of house repainting and atmospheric particulate concentration was shown in the third study by Michelson and Tourin.[5] A mailed survey of households in the upper Ohio River Valley established a linear relationship between repainting frequency and ambient levels of particulate matter. However, additional data are required to establish a more definite correlation. Maintenance data for additional cities should be added to the study to increase the study size; other factors to be considered include the effects of other pollutants that may be present and also the socioeconomic effects of the households in the survey.

In the fourth and final study, Watson and Jaksch[6] examined the soiling literature to determine which common household maintenance and cleaning tasks would be affected by atmospheric particulate soiling. The Watson and Jaksch study selected eight tasks as those most likely to be noticeably soiled by particulates; those tasks for which there was little or no evidence of being significantly affected by soiling were eliminated from consideration. The eight cleaning and maintenance tasks that would be affected by atmospheric particulate soiling are:

Indoor	Outdoor
Painting walls and ceilings	Painting walls
Wallpapering	Painting trim
Washing walls	Washing windows
Washing windows	
Cleaning venetian blinds	

No attempt was made, however, to determine the degree of the effect that soiling had on the frequency of the performance of these tasks; only that the effect would be significant.

Again, as was the case with the first two studies, the usefulness of the latter two studies is limited. No quantitative relationship between atmospheric particle concentration and cleaning frequency can be drawn. However, the evidence indicates that not only does suspended particulate cause soiling, but soiling affects the performance of cleaning and maintenance tasks. Thus, increased ambient particulate levels will lead to increased soiling, which will have a cost associated with its removal.

IV. Diesel Particulate Soiling

The previous descriptions of atmospheric particulate soiling refer to TSP (i.e., less than approximately 30 micrometers in diameter). Diesel particulate falls into a subclass of TSP (fine particulate, that are less than approximately 2.5 micrometers in diameter) and both its physical and chemical characteristics could quite likely cause it to have soiling properties different than those of TSP. Unfortunately, there exist very little direct experimental data demonstrating the relative soiling effect of fine particles or diesel particulate to those of TSP. Because of this, it is necessary to compare the characteristics of diesel particulate and TSP and postulate the effect of the differences on their overall soiling impact.

The physical and chemical properties of particulate which most affect the degree of soiling damage appear to be reflectance, stickiness, and size. Wallin has measured the optical reflectance of diesel particulate and found it to be generally about 3.5 times blacker than average urban particulate.[7] Thus, the change in reflectance due to deposition of diesel particulate will be greater than that of TSP because the difference between the reflectances of the surface (R_o) and the particulate (R_p) will be greater. This is caused by the high carbon content of diesel particulate, which has a reflectance of almost zero. Diesel particulate also appears to stick to surfaces more than the average particulate due to its oily nature (i.e., heavy hydrocarbons bound to the surface).[8]

In a report prepared for the California Air Resources Board, Sawyer and Pitz defined a "soiling index" as the ratio of the diesel particulate soiling to average urban particulate soiling on the basis of equal ambient mass concentrations.[8] They then went on to estimate the value of this index based on the relative properties of diesel particulate and TSP.

The effect of different optical properties was taken from Wallin's study and translated into an initial soiling index of 3 or 4 based on this single parameter. Because no experimental data are available on the stickiness of diesel particulate in quantitative terms, Sawyer and Pitz estimated a combined soiling index of 5 based on the combined effects of both reflectance and stickiness. To bracket the uncertainty, a range from 2.5 to 7.5 for diesel soiling indices was considered. (No effect due to different particle size was included. While it appears in some cases that small particles may deposit in greater amounts due to their greater diffusion capabilities, in other cases larger particles would deposit faster due to their greater mass. Thus, no clear preference based on size can be determined.)

As an example of how this soiling index would be used, one can assume an area with 75 ug/m^3 of TSP present. This concentration would have what could be called a soiling potential of 75 ug/m^3, since TSP is the base particulate (i.e., a one-to-one correspondence between mass concentration and soiling potential). If 5 ug/m^3 of diesel particulate were added to this atmosphere, the concentration of TSP would become 80 ug/m^3, an increase of 6.7 percent. However, using a soiling index of 5 for diesel particulate, the soiling potential with the addition of diesel particulate would be 100 ug/m^3 (75 + 5*5), an increase of 33 percent. Thus, one can see how adding a given concentration of diesel particulate to the atmosphere can have a much greater effect on soiling than would be indicated by its effect on particulate mass concentration.

V. Summary

Very little data are available on the effect of ambient particulate on the absolute degree of soiling and the frequency of cleaning. However, it is clear that ambient particulate, including diesel particulate, does result in soiling and has a cost associated with its removal. In addition, it appears that the degree of soiling associated with diesel particulate is greater than that of TSP on a mass concentration basis; possibly between 2.5 and 7.5 times as great. Thus, when relating the soiling effects of specific ambient concentrations of diesel particulate to those of TSP, the concentrations of diesel particulate should be increased by a factor substantially greater than 1 to place them in the proper perspective with the TSP concentrations.

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CHAPTER 8

ECONOMIC IMPACT

I. Introduction

A. Organization of Chapter

This chapter addresses the economic impact of the base scenario relative to the relaxed scenario. (Full descriptions of each scenario are given in the Introduction) The two basic trap-oxidizer designs and the associated regeneration systems are described in the remainder of this introduction.

The next two sections of this chapter examine the economic impact of particulate control on light-duty diesel vehicles (LDDVs) and trucks (LDDTs), and on heavy-duty diesel engines (HDDEs). The subsections in each section deal, in order, with estimating the cost of the hardware requirements for particulate control, examining the economic impact on affected vehicle and engine manufacturers, estimating the overall cost to the consumer of particulate control, and estimating the annual costs (for the years 1987 through 1995) and the 5-year aggregate costs (1987 through 1991 inclusive) of these controls.

B. Description of Trap Designs

The primary component of any system for the reduction of diesel particulate emissions is the trap-oxidizer. In addition to the trap itself, other hardware components are required, with the specific requirements depending on the basic trap-oxidizer design used. Trap-oxidizers (traps) can be broadly divided into categories on the basis of two factors: location or placement; and filter material.

An underfloor-mounted trap occupies approximately the same position, relative to the diesel engine, as is occupied by a catalytic converter on a gasoline-fueled vehicle. A close-coupled trap is located nearer to the engine, and is usually incorporated in the exhaust manifold design. Traps are also catalyzed or non-catalyzed, according to the presence or absence of catalytic materials to aid in the oxidation of accumulated particulate.

Detailed descriptions of the design and operation of each type of trap can be found in the EPA Trap-Oxidizer Feasibility Study.[1] For this economic analysis, the costs and economic impact are based only on the underfloor-mounted design, since it appears to be the preferred design of many trap-oxidizer and diesel vehicle/engine manufacturers. The possibility of close-coupled traps being used is addressed in Chapter 10. No clear preference for one of the two major filter materials has yet emerged; a brief description of each follows.

Although many filter materials have been investigated for use in traps, the current focus of development and testing is on ceramics and alumina-coated wire mesh. Ceramic traps utilize a non-catalytic, porous cordierite material [$2(MgO) + 2(Al_2O_3) + 5(SiO_2)$] for the substrate. This substrate is similar in construction to the support structure used for catalytic converters in gasoline-fueled applications, typically consisting of a honeycomb design with parallel square channels running the length of the unit. This trap design is being manufactured by Corning, NGK, and other firms.

Johnson-Matthey is the primary manufacturer of traps using alumina-coated wire mesh as the filter material. The form of the wire mesh trap is a long cylinder with a hollow central core. The exhaust flows radially through the mesh filter from the outside toward the hollow core. Catalytic coating of the wire mesh, lowering the temperature necessary for trap regeneration (oxidation of the accumulated particulate collected by the filter), is inherent in the Johnson-Matthey design.

Both types of trap are enclosed by a stainless steel shell, basically the same as that used for the exterior shell of a catalytic converter.

C. Description of Regeneration Systems

In addition, each type of trap requires a method of regeneration. Since excess accumulated particulate increases exhaust backpressure (thereby decreasing fuel economy and vehicle performance), it must be oxidized or burned off periodically. The temperature of the diesel exhaust stream is typically inadequate to initiate or sustain this oxidation. Therefore, a regeneration system is also required for effective particulate control.

The hardware components required for an effective regeneration system depend, in part, on whether the trap is catalyzed or non-catalyzed. The presence of catalytic material in the trap filter reduces the temperature increase needed for particulate oxidation, allowing the use of a less complex regeneration system than is required for non-catalyzed traps. Each of these is briefly described below; detailed explanations of the structure and functioning of trap regeneration systems are available elsewhere.[1,2]

A typical regeneration system for a non-catalyzed trap is based on a diesel fuel burner, which injects diesel fuel into the exhaust stream just before this stream enters the trap. Burning the added fuel increases the exhaust temperature enough to ignite the accumulated particulate. The engine exhaust flow is temporarily routed around the trap, while the burner and trap are supplied with a controlled air flow to ensure continued oxidation of the trapped particulate without excessive heating.

This typical regeneration system has seven primary hardware components: a burner head, a fuel delivery system, an ignition system, an auxiliary combustion air system, an exhaust diversion system, system control sensors, and an electronic control unit (ECU).

The burner head provides a location for mounting the fuel spray nozzle, ignition plug, and auxiliary air nozzle. It is also assumed to include a gas distribution baffle for evenly distributing the combustion products over the cross-section of the trap.

The fuel delivery system provides the diesel fuel necessary for initiating the trap regeneration process. This system includes a fuel spray nozzle, a fuel feed line, and a fuel solenoid valve.

The fuel ignition system may be one of two basic types. One system consists of a long-life spark plug, a step-up voltage transformer, and signal conditioning electronics for generating a high-voltage discharge. An alternative to this system is a glow plug, like those used to cold-start diesel engines.

The auxiliary air combustion system, which provides a controlled air supply to the burner and trap to sustain the oxidation of the accumulated particulate, consists of an air pump, a check-valve, a diverter valve, and an air delivery line. The check-valve prevents exhaust backflow into the air pump, while the diverter valve provides an alternate path in the event that combustion air must be diverted from the filter. The air delivery line connects the air pump to the burner head.

The exhaust diversion system temporarily reroutes the engine exhaust stream around the trap during the regeneration process. It consists of a vacuum motor driven by the ECU and an engine-driven vacuum pump, which generates the vacuum required for operation of various control elements. An alternative to this system, not requiring a vacuum pump, is a solenoid valve operator.

System control sensor requirements include two temperature sensors, and a control sensor for determining the need for trap regeneration. Temperature sensors are required for detecting overheating in the trap filter during the regeneration, and for ensuring that the engine has attained normal operating temperature before the regeneration is initiated. The sensor determining the need for trap regeneration could be either an engine revolution or vehicle mileage timer, or an exhaust backpressure sensor.

The most important regeneration system component, in terms of system control, is the ECU. The ECU interprets signals received from the various sensors in order to maintain control of the regeneration process.

As was noted earlier, the regeneration system for a catalyzed trap can be less complex, since the increase in the exhaust stream temperature required is much smaller. Of the seven primary hardware components required for the non-catalyzed trap regeneration system described above, only the system control sensors and the ECU are needed in basically the same form for the catalyzed trap regeneration system.

Some type of auxiliary air combustion system is still required;[1] however, since the exhaust flow through the catalyzed trap is maintained during the regeneration process, a reed valve system may be adequate. The burner head, exhaust diversion system, fuel delivery system, and ignition system described above are not required.

However, an alternate system for providing a moderate rise in the temperature of the exhaust stream is still required. One such method, which has been successfully tested on a Volkswagen Rabbit in conjunction with the Johnson-Matthey catalyzed wire-mesh trap, is known as delayed in-cylinder fuel injection. A small amount of fuel is injected into the cylinder during the exhaust stroke, when the cylinder is too cool to ignite the fuel. The injected fuel is carried in the exhaust stream to the catalyzed trap, where it is ignited. Since the existing fuel system is used to inject the fuel, the only hardware necessary is a mechanism for transferring a portion of the fuel being metered from a "normal" injector to the "delay" injector.

Though actually not part of the trap or of the regeneration system, one other vehicle modification affecting the exhaust system should be discussed here. The exhaust pipe, leading from the engine to the trap-oxidizer, will have to be fabricated of stainless steel. If fabrication of this pipe using normal steel were continued, periodic replacement would be required, greatly increasing the chances of the trap-oxidizer being removed from the vehicle. This modification is required for all of the trap designs and regeneration systems discussed.

II. Light-Duty Diesels

A. Introduction

This section examines the impact of particulate control for LDDVs and LDDTs. Since the methodology and many of the basic assumptions used in this analysis are the same for both light duty and heavy duty, this section contains considerably more detail than does the next section on heavy-duty diesels.

The next subsection estimates the costs, in terms of the retail price equivalent (RPE), of each of the basic trap designs and regeneration systems described in the introduction. These costs are largely a function of the size (volume) of the trap-oxidizer. This discussion is followed by subsections treating the economic impact on diesel manufacturers, the overall cost to the consumer, and the annual and 5-year aggregate costs of these controls.

After the costs of the hardware (trap-oxidizer and regeneration system) are estimated, the subsequent analysis examines the economic impact under two regulatory scenarios (base and relaxed), and under two sets of future diesel sales projections ("best estimate" and "worst case"). The regulatory scenarios are described in detail previously. The best estimate sales projections[3] are exactly what the designation implies, while the worst case sales projections are based on the maximum increases in diesel sales that appear to be reasonable. (The term "worst case" refers to the impact of increased diesel sales on total particulate emissions, and the resulting environmental effects.)

The cost of the two basic trap-oxidizers, catalyzed and non-catalyzed, were previously estimated in the Regulatory Analysis that accompanied the original light-duty particulate control regulations.[4] The model used to estimate the manufacturing costs of each trap design, which was developed by Lindgren,[5] is again used in this analysis, with cost estimates provided by the trap manufacturers incorporated into the model where available. The Lindgren model for estimating the RPE of manufacturing costs[5] is based on the application of adjustment factors to the estimated sum of direct material and labor, and fixed overhead costs. These factors are expressed as 1.0 plus the sum of the adjustment terms, as shown below:

$$RPE = [(DM+DL+OH)(1+CA+SP)+TE+LBE](1+CA+CP+DP)+RD+TE \quad (1)$$

Where:

DM = Direct material cost.
DL = Direct labor cost.
OH = Fixed and variable overhead.
CA = Corporate allocation term of adjustment factor.
SP = Supplier profit term of adjustment factor.
TE = Tooling expense.
LBE = Land and building expense.
CP = Corporate profit term of adjustment factor.
DP = Dealer overhead and profit term of adjustment factor.
RD = Research and development cost.

Some of the values used in equation 1 were taken directly from Lindgren's work,[5] while others have been adjusted based on more recent analyses.[4] Additional adjustment factors for inflation and production volume (i.e., economy of scale) are also incorporated in this analysis. These are described in more detail below.

Regeneration system costs have also been estimated in the past.[4] In this analysis, these earlier estimates are essentially supplanted by more recent work performed by Mueller Associates[2] under an EPA contract.

B. Trap-Oxidizer System Costs

1. Introduction and Assumptions

The adjustment factors for inflation and production volume are independent of the trap design or regeneration system used. Therefore, these are discussed first, before estimating the specific manufacturing costs for each case.

Some of the manufacturing cost data that went into the development of Lindgren's model and into the previous EPA analyses dates from as early as 1978. Therefore, an adjustment factor for inflation must be determined. For application to particulate control hardware, the increase in LDV (new car) prices from 1978 through 1982 appears to be a more appropriate estimate of inflation than the rise in the Consumer Price Index over the same time span. New car prices were 33 percent higher in 1982 than in 1978, with annual increases of 6.2, 7.4, 7.5, 6.8, and 1.6 percent in 1978, 1979, 1980, 1981, and 1982, respectively.[6] These inflation rates are used in this analysis.

Production volumes of different traps by various trap manufacturers are uncertain. The assumption in this analysis is that two manufacturers will supply trap-oxidizers, and that each of them will supply approximately half of total production.

It is also necessary to distinguish between different sizes of LDDVs and LDDTs, since the size (volume) of the trap is dependent on the engine size (displacement). In this analysis, LDDVs are divided into small, medium, or large on the basis of engine displacement. Small LDDVs are those equipped with 1.6- to 1.8-liter (L) engines, medium LDDVs are equipped with 2.0L to 2.8L engines, and large LDDVs are those with 3.0L and larger engines. It is assumed that the projected LDDV sales will be divided approximately equally among these three size classes. The LDDTs are considered to be either small (engines under 4.3L) or full-size (4.3L and larger engines). Full-size LDDTs are assumed to sell at a 4:1 ratio to small LDDTs.

Based on the best estimate LDDV sales projections[3] and the projected rates of trap usage (see Chapter 1), a standard average production level of 200,000 traps annually for years 1987-91, for each of the three LDDV size classes, is a reasonable estimate. This estimate would increase if based in worst case sales projections.

Projected average annual sales of small LDDTs are approximately half those projected for each LDDV size class (see Chapter 1). Thus the standard average trap production level for small LDDTs is estimated at 100,000 annually, or half of the standard production level for each LDDV size class. Full-size LDDT sales are projected to be roughly twice those for each LDDV size class, therefore, the standard average production level of traps for full-size LDDTs is estimated to be 400,000.

In order to develop adjustment factors based on the standard average production levels of traps for each of the five size classes of light-duty diesels, the "learning curve" must be known. For trap-oxidizer production, the learning curve is assumed to be 12 percent.[4] The learning curve concept is applied to the production levels by first assuming that some standard average production level serves as a baseline, for which the production level adjustment factor is equal to one (i.e., no adjustment). Under a 12 percent learning curve, doubling the baseline production level leads to a 12 percent decrease in per-unit manufacturing costs, or an adjustment factor of 0.88. Conversely, halving the baseline production level leads to a 13.6 percent increase in per-unit manufacturing costs, expressed as an adjustment factor of 1.136 (1.0/0.88).

Application of the learning curve to the production of trap-oxidizers is done by assuming that the baseline standard average production level is 200,000 traps annually, the level estimated for each of the three LDDV size classes. Therefore, no adjustment factor for the level of production is applied to the 200,000-trap annual production level assumed for each LDDV size class. The adjustment factor for full-size LDDTs is 0.88, representing a 12 percent decrease in per-unit manufacturing costs resulting from a doubling of LDDV trap production. For small LDDTs the production level adjustment factor is 1.136, reflecting the production level of small LDDT traps being half that of each LDDV size class.

The production volumes of traps for each size-based class of light-duty diesels are all based on the best estimate sales projections. Under the worst case projections, LDDV production would double and LDDT production would increase 50 percent over the best estimate projections. The effect would be to lower per-unit trap manufacturing costs by 12 percent for LDDVs and by 6.2 percent for LDDTs.

With the information given above, the discussion can now be focused on estimating the manufacturing costs of each trap design and of the regeneration systems.

2. Non-Catalyzed (Corning) Trap

A formula for determining the manufacturing costs of non-catalyzed traps has been developed and used in previous EPA analyses.[1,7] This formula was derived by relating the various trap components to similar or identical components of a monolithic catalyst for gasoline-fueled engines, for which confirmed manufacturing costs are already known. After applying the adjustment factors, including those for inflation and production volume (based on the best estimate sales projections) determined in the preceding section, the formulae are:

For LDDVs:

$$\text{RPE} = \$23 + 0.318(V) \qquad (2A)$$

For small LDDTs:

$$\text{RPE} = \$26 + 0.356(V) \qquad (2B)$$

For full-size LDDTs:

$$\text{RPE} = \$20 + 0.280(V) \qquad (2C)$$

Where:

V = the volume of the trap, in cubic inches.

As an example of applying these equations, consider the case of a non-catalyzed trap that was recently tested successfully, by Southwest Research Institute, on a Mercedes-Benz 300D. This trap had a volume of 302 cubic inches (5.66 inch diameter x 12 inch length); substituting 302 for V in equation 2A (for LDDVs) leads to an estimated manufacturing cost of about $119.

As mentioned earlier, traps of various sizes (volumes) will be fitted to different sizes of engines. Trap size can logically be expected to be a function of volumetric exhaust flow of the engine.[1] While data on the typical volumetric exhaust flows of various engines are not readily available, fuel consumption (the inverse of fuel economy) is an adequate surrogate measure.[1] The ratios of the fuel consumptions, or the inverse ratios of the fuel economies, over the FTP (EPA urban) driving cycle can be used to extrapolate trap volume requirements for other engine sizes, given a known reference point: The Mercedes-Benz 300D mentioned above has an EPA city fuel-economy rating of 26 miles per gallon (mpg).

Projected fuel economies for each of the five size classes under consideration, in 1990, are given in the table below:

Size Class	Engine	Projected FE
Small LDDVs	1.6 to 1.8L	51.2 mpg
Medium LDDVs	2.0 to 2.8L	43.9 mpg
Large LDDVs	3.0L and up	37.8 mpg
Small LDDTs	under 4.3L	44.8 mpg
Full-size LDDTs	4.3L and up	33.6 mpg

These estimates were derived from fuel economy estimates for gasoline engines in 1990,[3] with a 25 percent improvement assumed in diesel engine fuel economy over the corresponding gasoline engines.

Using the Mercedes 300D (26 mpg fuel economy, 302 cubic inches trap volume) as the reference point, and applying the fuel consumption ratios as discussed above, the resulting trap volume requirements are:

Size Class	Trap Volume
Small LDDVs	153 cubic inches
Medium LDDVs	179 cubic inches
Large LDDVs	208 cubic inches
Small LDDTs	175 cubic inches
Full-size LDDTs	234 cubic inches

These volumes can be substituted for V in the equations 2A-2C, yielding estimated manufacturing costs of $72, $80, and $89 for small, medium, and large LDDVs, and $88 and $87 for small and full-size LDDTs, respectively.

As shown in the table of trap volumes above, the small LDDT trap is projected to require a volume only 4 cubic inches less than that of the medium LDDV trap. The medium LDDV trap is also estimated to cost less to manufacture than the small LDDT trap, $80 versus $88. Thus it is more economical to produce one trap, of the size required for medium LDDVs, for both medium LDDV and small LDDT applications. Combining the standard average production levels of 200,000 annually for medium LDDVs and 100,000 annually for small LDDTs into a new standard average production level of 300,000 traps, the assumed 12 percent learning curve lowers the per-unit cost to $74.

The manufacturing costs presented above are summarized in Table 8-1. Confidential estimates of manufacturing costs supplied by Corning, while not firm, indicate that the estimates shown in Table 8-1 are reasonably accurate.

Under the worst case sales projections, the cost estimates given above are reduced by 12 percent for LDDVs and by 7.2 percent for LDDTs. These estimates are also shown in Table 8-1.

3. Catalyzed (Johnson-Matthey) Trap

In the Regulatory Analysis for the 1985 light-duty diesel particulate regulations,[4] the cost of a catalyzed trap was also estimated by relating the components of the trap to similar or identical components of a monolithic, ceramic catalytic converter, with washcoat and noble metals included. A formula was then developed for estimating the manufacturing cost based on the trap volume.

Johnson-Matthey has since publicly stated that the manufacturing cost of their catalytic trap substrate, ready for canning, was $100 (in 1982 dollars) for a trap intended for use with a 2.0L engine. If the volume of this trap is assumed to be equal to that of a trap recently tested successfully on a Volkswagen Rabbit with a 1.6L engine (345 cubic inches), then

Table 8-1

Light-Duty Trap Costs (1983 dollars)

Best Estimate Sales Projections

Vehicle Class	Non-Catalyzed Trap	Catalyzed Trap
Small LDDVs	$72	$188
Medium LDDVs	$74	$199
Large LDDVs	$89	$246
Small LDDTs	$74	$199
Full-Size LDDTs	$87	$236

Worst Case Sales Projections

Vehicle Class	Non-Catalyzed Trap	Catalyzed Trap
Small LDDVs	$63	$165
Medium LDDVs	$66	$178
Large LDDVs	$78	$216
Small LDDTs	$66	$178
Full-Size LDDTs	$81	$219

the RPE of the manufacturing cost can be determined using the Johnson-Matthey cost information. The $100-estimated manufacturing cost must first be inflated to 1983 dollars and then substituted for the non-catalyzed substrate manufacturing cost in equations 2A-2C. The effects on the total cost of canning, corporate overhead and profit, and dealer mark-up are assumed to be unchanged from the non-catalyzed trap.

It is assumed that the fixed costs (i.e., tooling and machinery, fixed overhead) are the same for both trap types, meaning that all variable costs can be expressed as a function of trap volume. Finally, by combining the production of traps for medium LDDVs and small LDDTs as was discussed in the preceding section, the following equations result:

For small and large LDDVs:

$$\text{RPE} = \$23 + 0.582(V) \qquad (3A)$$

For medium LDDVs and small LDDTs:

$$\text{RPE} = \$22 + 0.536(V) \qquad (3B)$$

For full-size LDDTs:

$$\text{RPE} = \$20 + 0.501(V) \qquad (3C)$$

Where:

V = the volume of the trap, in cubic inches.

The 345 cubic inch catalyzed trap mentioned above is estimated to cost about $224, based on equation 3A. Use of the methodology developed in the original Regulatory Analysis,[4] with adjustments made for inflation, production volume, and more recent precious metal costs, yields an estimated cost of $212 for a 345 cubic inch catalyzed trap. Thus incorporating the Johnson-Matthey estimate into the equations 2A-2C changes the estimated overall trap cost by less than 6 percent.

As in the non-catalyzed case, it must be assumed that traps of different sizes (volumes) will be produced for use with different engines. The 1990 estimated fuel economies for light-duty diesels given above are used here, with the reference point changed to the Volkswagen Rabbit (42 mpg fuel economy, 345 cubic inch trap volume). Applying the ratios of fuel consumption as a surrogate measure of volumetric exhaust flow, as was done in the non-catalyzed case, yields the following trap volume requirements:

Size Class	Trap Volume
Small LDDVs	283 cubic inches
Medium LDDVs and small LDDTs	330 cubic inches
Large LDDVs	383 cubic inches
Full-size LDDTs	431 cubic inches

Substituting these volume requirements for V in equations 3A-3C gives the RPE of the manufacturing cost. The estimated costs based on the equations are: $188 for small LDDV traps, $199 for medium LDDV and small LDDT traps, $246 for large LDDV traps, and $236 for full-size LDDT traps. These cost estimates for catalyzed traps are also summarized in Table 8-1.

Equations 3A-3C and the cost estimates above are based on the best estimate sales projections. The impact of the "worst-case" sales projections on these estimates is the same as on the non-catalyzed cost estimates, with the LDDV costs reduced by 12 percent and the LDDT costs reduced by 6.2 percent from the figures above. These estimates are also shown in Table 8-1.

4. Regeneration System Costs

The main components of trap regeneration systems, for both catalyzed and non-catalyzed trap-oxidizers, were described in the introduction. The positive regeneration system for non-catalyzed traps, which actively initiates the burn-off of the accumulated particulate by injecting ignited diesel fuel into the exhaust stream, is dealt with first. The costs estimated for both regeneration systems are largely based on the analysis performed by Mueller Associates for EPA.[2]

The components of both types of regeneration system are listed, with the estimated RPE of the manufacturing costs, in Table 8-2. These estimates are based on the production levels corresponding to the best estimate sales projections. Since almost all of the regeneration system components listed in Table 8-2 are also manufactured for purposes other than particulate control, the production levels are higher than those of trap components. On this increased base production level, the impact of the "worst-case" sales projections is much smaller. Thus, any changes in these estimates due to increases in trap-equipped diesel sales are also much smaller, and are not shown in Table 8-2.

The hardware components for each type of regeneration system are listed in Table 8-2, and discussed below, in the same order as they were described in the introduction.

Table 8-2

Light-Duty Regeneration System Costs (1983 dollars)

Hardware Item	Retail Price Equivalent
Non-Catalyzed Trap:	
Burner Head	$7
Fuel Delivery System	$9
Fuel Ignition System*	$5-26
Auxiliary Combustion Air System	$30
Exhaust Diversion System*	$11-14
System Control:	
Sensors	$12
ECU**	$10
Subtotal	$84-108
Stainless Steel Exhaust Pipe	
Small and Med. LDDVs, small LDDTs	$16
Large LDDVs, full-size LDDTs	$27
Total System Cost	$100-135
Catalyzed Trap:	
Delayed In-Cylinder Fuel Injection Mechanism	$15
Auxiliary Combustion Air System (Reed Valve)	$6
System Control:	
Sensors	$12
ECU**	$10
Subtotal	$43
Stainless Steel Exhaust Pipe	
Small and Med. LDDVs, small LDDTs	$16
Large LDDVs, full-size LDDTs	$27
Total System Cost	$59-70

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* Explanation of cost ranges appear in the text.

** Derivation of the ECU cost appears in the text.

The burner head is assumed to be fabricated of stamped and welded Type 409 stainless steel, and has an estimated cost of $7.[2] The fuel delivery system, for supplying the fuel to be ignited to initiate the regeneration process, has an estimated cost of $9.[2]

Two basic fuel ignition systems were described in the introduction. The more costly system (long-life spark plug, step-up voltage transformer, and signal conditioning electronics) is estimated to cost $26.[2] While the use of a glow plug is less expensive, with an estimated cost of $5,[2] it is also less reliable for ignition when the temperature of the exhaust stream is relatively low. Both of these options are included in Table 8-2.

The regeneration process requires a controlled supply of air to the burner and trap to sustain particulate oxidation. The total cost of the auxiliary air combustion system (pump, delivery line, and valves) is estimated to be $30.[2]

Exhaust must temporarily be rerouted around the non-catalyzed trap during regeneration. The exhaust diversion system for accomplishing this is estimated to cost between $11 and $14.[2] The lower cost is for a system utilizing a vacuum motor (an engine-driven vacuum pump is assumed to already be present on the vehicle), while a system using a solenoid valve operator is represented by the higher cost (no vacuum pump need be present on the vehicle).

System control requires the use of several sensors. The estimated costs are $9, for a sensor to detect overtemperature in the filter during the regeneration, and $1, for a sensor to ensure that the engine has attained the proper operating temperature before regeneration is initiated.[2] The sensor for determining the need for trap regeneration could be either an engine revolution or vehicle mileage timer, or, an exhaust backpressure sensor. The cost of the former is negligible,[2] while the latter is estimated to cost no more than $2.[8] Since the backpressure sensor is more desirable, however, the latter estimate is included in Table 8-2.

The critical system control component is the electronic control unit (ECU). For gasoline-fueled engines, the current total cost of an ECU is approximately $75.[2] Several factors make this cost inappropriate for direct use in Table 8-2. First, the current ECU is typically much more sophisticated than is needed for regeneration system control. Second, it is highly probable that ECUs on diesels after 1987 will serve several purposes in addition to their emission control functions. (For example, Isuzu's 1983 diesel vehicles contain an ECU which functions to improve fuel economy and vehicle performance, as well as to control dashboard lighting and other miscellaneous devices or "gadgets.") Third, and most importantly, continuing advances in microprocessor technology can be expected to further reduce the cost of ECUs in constant-dollar terms, while simultaneously widening the scope of potential automotive applications.

No data are available on manufacturers' plans for the installation of ECUs to serve functions other than emission control. A conservative estimate is that half of all LDDVs and LDDTs will be equipped with ECUs, for reasons other than emission control, during the period 1987-95. The remaining half of LDDV/LDDT production would incorporate ECUs in order to comply with emission control requirements; however, once incorporated into the vehicle design they would certainly serve additional valuable functions.

The ECU in the LDDV or LDDT of the future will have four primary functions: improving fuel economy, improving vehicle performance, device and "gadget" control, and emission control. Allocating one-quarter of the total $75 cost to the emission control aspects of the ECU gives an estimated cost of approximately $19 due to particulate control.

Assuming that half of the ECUs installed for emission control will be required solely for particulate control reduces the fleetwide average per-vehicle cost to $10. If ECUs are installed in more diesel vehicles than projected for purposes of NOx control, this estimate may be reduced even further.

This regeneration system has a total estimated RPE of between $85 and $109, depending mostly on the fuel ignition system chosen. As discussed in the introduction, a stainless steel exhaust pipe will also be required for trap-equipped vehicles. When a credit for the deleted standard steel exhaust pipe is included, the additional cost of this modification is estimated as $16 (for small and medium LDDVs and small LDDTs) to $27 (for large LDDVs and full-size LDDTs). These costs are also shown in Table 8-2.

The regeneration system for a catalyzed trap should be less complex than the system required for non-catalyzed traps, as was explained in the introduction. While detailed cost estimates such as those given above are not available for this simpler system, the savings over the "burner system" can be estimated using the information in Table 8-2.

No burner head assembly is required. The fuel delivery system is replaced by a mechanism for transferring a small amount of fuel from the normally-functioning injector to the "delay" injector. This mechanism is expected to cost about $15.[2] The auxiliary air combustion system described above can be replaced by a reed valve (estimated cost $6),[2] since the continued exhaust flow through the catalyzed trap during regeneration will provide the required suction.

The sensors and the ECU, required for regeneration system control, are basically identical for either system. The stainless steel exhaust pipe is also required for both systems. The cost estimates for these components of the catalyzed trap regeneration system are the same as for the non-catalyzed case. All of this information is also shown in Table 8-2.

5. Total Trap-Oxidizer System Costs

The total cost of the trap-oxidizer system is the sum of the costs estimated for the trap and for the regeneration system. Summaries of these costs under both the best estimate and the worst case diesel sales projections are shown in Table 8-3. Since the widths of the ranges in cost are quite small, relative to the absolute costs, only the midpoints of the cost ranges are shown in Table 8-3. These "midpoint" estimates are used throughout the rest of the analysis.

It is clear from Table 8-3 that, despite the savings associated with the regeneration system, the total cost of the catalyzed trap-oxidizer system is still estimated to be substantially more than that of the non-catalyzed system. Since it is considerably less expensive, and appears to be the preferred design of most diesel manufacturers, only the cost of the non-catalyzed trap-oxidizer system is used in the remainder of this analysis.

C. Economic Impact on Diesel Manufacturers

In this section, the impact of the base scenario on manufacturers' light-duty diesel sales, capital investments and cash flow will be analyzed. Only the costs of the trap-oxidizer system and the associated fuel economy penalty are considered here. There are no test facility costs associated with the base scenario. Certification costs were shown to be negligible in the Regulatory Analysis to the 1985 particulate standards.[4]

Table 8-3

Total Light-Duty Trap-Oxidizer System Costs (1983 Dollars)

Best Estimate Sales Projections

Vehicle Class	Non-Catalyzed Trap	Catalyzed Trap
Small LDDVs	$185	$246
Medium LDDVs	$187	$258
Large LDDVs	$213	$316
Small LDDTs	$187	$258
Full-Size LDDTs	$211	$306

Worst Case Sales Projections

Vehicle Class	Non-Catalyzed Trap	Catalyzed Trap
Small LDDVs	$176	$224
Medium LDDVs	$179	$237
Large LDDVs	$202	$286
Small LDDTs	$179	$237
Full-Size LDDTs	$205	$289

1. Impact on Manufacturer's Sales

The impact of the base scenario on light-duty diesel sales depends primarily on three factors. First is the vehicle price increase resulting from the additional cost of installing a trap-oxidizer. Second is the fraction of vehicles requiring trap-oxidizers, which was determined in Chapter 1. Third is the 1-3 percent fuel economy penalty associated with the use of trap-oxidizer technology.[9]

The next step is applying these factors to determine a net impact on future diesel sales. This has already been done for a number of potential combinations of trap costs and trap usage rates, in a study performed by Jack Faucett Associates (JFA) for EPA[3] using consumer information on diesel vehicle purchases from Chase Econometrics.[10] JFA estimated the impact on LDDV and LDDT sales assuming trap-oxidizer costs of $300, $500, and $800, and trap usage rates of 0 percent, 35 percent, 65 percent, and 90 percent. An average fuel-economy penalty of 2 percent for vehicles equipped with traps was also incorporated. It was assumed that the largest diesel vehicles would be equipped with traps first, the medium-size diesels next, and the smallest diesels last, until the overall trap usage rate was met.

To simplify the application of JFA's results, an average trap-oxidizer cost will be used for each vehicle class (LDDV and LDDT). The trap-oxidizer costs for each vehicle size will be weighted by the relative sales of each vehicle size, as estimated by JFA.[3] This average cost is $213 for LDDVs, and $219 for LDDTs.

Using the average trap system costs and the trap penetration rates, the future sales of light-duty diesel vehicles and trucks can be projected by interpolation of the JFA estimates. Table 8-4 shows the projected sales for the "relaxed scenario," where no traps are required on light-duty diesels, under both best estimate and "worst-case" diesel sales projections. These figures represent the maximum number of vehicles projected to be sold, as these vehicles do not bear the cost of a trap-oxidizer.

Also shown are the effects of the base scenario on light-duty diesel sales under both the best estimate and "worst-case" sales projections. As can be seen, the impact of the base scenario is greatest in the early years (e.g., 1987) and diminishes with time. In addition, LDDV sales are affected more than LDDT sales. The largest impact occurs in 1987, when 30,000 LDDV sales are lost (3.4 percent of total LDDV sales under the best estimate sales projections). By 1995, this loss

Table 8-4

Light-Duty Diesel Sales Projections* (in thousands)

	Sales and Percent Reduction		
	1987	1990	1995
Best Estimate Sales Projections			
Relaxed Scenario			
LDDV Sales	912	1,300	1,380
LDDT Sales	714	1,029	1,322
Total: LDDVs and LDDTs	1,626	2,329	2,702
Base Case Scenario			
LDDV Sales	881 (3.4%)**	1,260 (3.2%)	1,355 (1.8%)
LDDT Sales	704 (1.4%)	1,018 (1.0%)	1,310 (0.9%)
Total: LDDVs and LDDTs	1,585 (2.5%)	2,278 (2.2%)	2,665 (1.3%)
Worst Case Sales Projections			
Relaxed Scenario			
LDDV Sales	1,824	2,875	3,600
LDDT Sales	952	1,400	2,340
Total: LDDVs and LDDTs	2,776	4,275	5,940
Base Case Scenario			
LDDV Sales	1,793 (1.7%)**	2,835 (1.4%)	3,575 (0.7%)
LDDT Sales	942 (1.0%)	1,389 (0.8%)	2,328 (0.5%)
Total: LDDVs and LDDTs	2,735 (1.5%)	4,224 (1.2%)	5,903 (0.6%)

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* California sales included.

** Percent reduction in sales from relaxed scenario.

diminishes to 25,000 on a much larger sales base (1.8 percent of total "best estimate" LDDV sales). Losses of LDDT sales are roughly one-third to one-half as great, in the range of 10,000 to 12,000 units annually.

An underlying assumption of this analysis is that the manufacturers will pass the total cost of the trap-oxidizer system on to the consumer. Manufactuers have been selling diesel vehicles at a premium to consumers willing to pay extra for ownership of a relatively new and advantageous product.[3] Thus diesel manufacturers have been generating higher than normal profits on diesel sales, relative to profits on comparable gasoline-fueled vehicles. If this situation continues, then manufacturers might be able to absorb some of the costs of a trap-oxidizer system by reducing their above-normal profit margin. However, it is expected that the premium in the price being paid for diesels will decrease as increased competition from other diesel manufacturers brings profit margins down to normal levels. Manufacturers then would not be able to absorb the trap-oxidizer cost, and would pass it through to the consumer.

Even if diesel sales decrease as a result of manufacturers adding trap-oxidizer costs to their vehicle sales prices, it is unlikely that the automobile industry as a whole would lose a sale. JFA concluded that any consumer deciding not to buy a diesel would purchase a gasoline-fueled vehicle instead. This finding is not surprising when it is considered that the functions of the two types of vehicles are nearly identical, and that only the economics of ownership differ. Thus the automobile industry as a whole should suffer no lost sales due to the base scenario particulate control standards.

2. Investment Costs and Cash Flow Effects

Two other effects that emission regulations can have on diesel manufacturers are increasing required capital investment (i.e., tooling, machinery, research and development (R&D), etc.) and reducing cashflow. These effects are examined below.

The bulk of the capital investment associated with the required use of trap-oxidizers is not expected to be borne by diesel vehicle/engine manufacturers, but rather by the manufacturers of emission control equipment, such as Corning, NGK, and Johnson-Matthey. The catalyst manufacturers already have developed the necessary substrate technology, and the diesel manufacturers have shown little interest in this area. Even though the manufacturers of emission control equipment will have to finance the necessary investments, they all have indicated their willingness and ability to enter this market. Thus pre-production investment costs should not be a problem for any affected entities.

With respect to other investment costs, light-duty manufacturers are presently incurring some R&D costs associated with applying trap-oxidizer technology to their vehicles. However, much of this work has already been completed, and future R&D should be no less fundable. Thus, R&D and capital investment requirements should not have a significant adverse impact on any manufacturers' investment plans.

Given the above, the only impact on cash flow will result from the inventory of traps, individually and on partially manufactured vehicles. The time each trap is held should be much shorter than that for an entire vehicle, which averages about 90 days.[4] This turnover period should be short enough to not significantly affect a manufacturer's cash flow. For example, assuming an average turnover time of six weeks and industry-wide sales of 1.5 million LDDVs and LDDTs, the value of the trap-oxidizers on hand at any given time would only be $37.5 million. This is less than $4 per vehicle spread across total light-duty sales.

D. Total Cost to the Consumer

The bulk of the total consumer cost of particulate control is the increased "sticker price" of an LDDV or LDDT. Assuming that the full RPE of manufacturing cost is passed through to the retail purchaser, the entries of Table 8-3 represent both total trap-oxidizer system costs and the increase in new LDDV/LDDT purchase prices due to particulate control. The remainder of the total cost to the consumer results from the fuel-economy penalty, and from any increases in the maintenance costs for light-duty diesels resulting from the addition of trap-oxidizer systems.

Installation of trap-oxidizer systems is expected to result in an average fuel economy penalty of 2 percent.[9] Estimating the cost of this penalty over the life of an LDDV/LDDT requires that the following information be specified: cost of diesel fuel, discount rate, average vehicle miles travelled (VMT) for LDDVs and LDDTs, average LDDV and LDDT fuel economy, and average LDDV and LDDT lifetime. In this analysis, the assumptions used are: $1.20/gallon for the average cost of diesel fuel; a 10 percent discount rate; average annual VMT of 10,000 for LDDVs and 10,800 for LDDTs; the estimated 1990 fuel economies for each size class that were used in determining trap volume requirements; and average lifetimes of 10 years for LDDVs and 11 years for LDDTs.[11,12]

To calculate the cost of the fuel economy penalty, the estimated 1990 fuel-economy values (II.B.2.) are reduced by 2 percent. Knowledge of the fuel economy and annual VMT allows the annual fuel consumption to be determined, which then is multiplied by $1.20/gallon to yield annual fuel costs. The 10 percent discount rate and the lifetime periods are used to determine the present value of lifetime fuel expenditures in the year of vehicle purchase.

The process is repeated without including the 2 percent fuel-economy penalty, and subtraction of the lower total from the higher total gives the cost of the fuel-economy penalty to the consumer. Carrying through these calculations, the net present value of the fuel-economy penalty in the year of vehicle purchase is $33 for small LDDVs, $46 for medium LDDVs, $52 for large LDDVs, $41 for small LDDTs, and $55 for full-size LDDTs.

Increased maintenance costs will result only from maintenance of the trap regeneration system, since the trap itself is expected to be maintenance-free and use of the trap-oxidizer system will have no adverse impacts on other vehicular maintenance requirements. Regeneration system maintenance is likely to be limited to replacement of the one or both of the temperature sensors used for system control. This maintenance is estimated to require about one hour labor and $10 in new parts, and should only be required once during the lifetime of the vehicle. Assuming a labor charge of $25 per hour, the total cost of this maintenance is $35. This maintenance should occur approximately halfway through the lifetime of the vehicle, or about five years after the initial purchase. Discounted to the year of the vehicle purchase, the regeneration system maintenance cost is estimated to be $22.

Use of trap-oxidizer systems will reduce the cost to the consumer for exhaust system maintenance. By using a stainless steel exhaust pipe (to discourage in-use trap removal), the need for periodic replacement of the standard steel exhaust pipe is eliminated. A conservative estimate of one exhaust pipe replacement, at roughly the midpoint of the vehicle lifetime (5 years), being eliminated results in consumer savings of $21 (for small LDDVs and LDDTs, and medium LDDVs) to $36 (for large LDDVs and full-size LDDTs).[4] As in the estimated cost of regeneration system maintenance, a 10 percent discount rate is assumed.

The sum of the increased LDDV/LDDT initial purchase price, the cost of the fuel-economy penalty, and the cost of regeneration system maintenance, less the savings on exhaust system maintenance, represents the total cost to the consumer of particulate control. These costs are summarized in Table 8-5 for each of the five size classes of light-duty diesels, and range from $210 to $266 per vehicle. Against the net

Table 8-5

Total Cost to Consumers of Owning and Operating a Light-Duty Diesel Equipped With a Trap-Oxidizer (1983 dollars)*

	Small LDDVs	Medium LDDVs	Large LDDVs	Small LDDTs	Full-Size LDDTs
Trap-Oxidizer System:					
Best Estimate Sales Projections	$185	$187	$213	$187	$211
"Worst-Case" Sales Projections	$176	$179	$202	$179	$205
Maintenance Costs	$22	$22	$22	$22	$22
Maintenance Savings	($21)	($21)	($36)	($21)	($36)
Cost of Fuel Economy Penalty	$33	$46	$52	$41	$55
Total Cost to Consumer:					
Best Estimate Sales Projections	$219	$234	$266	$229	$252
"Worst-Case" Sales Projections	$210	$226	$255	$221	$246
Total Cost of Owning and Operating Vehicle[13]	$14,168	$16,377	$19,418	-	-
Cost Increase Due to Trap-Oxidizer	1.5%	1.5%	1.4%	-	-

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* All costs are discounted to year of vehicle purchase using a 10 percent discount rate.

present value of the cost of owning and operating an LDDV over its lifetime,[13] these costs represent increases of 1.4 to 1.5 percent.

E. Annual Costs

The annual costs of the base case scenario are shown, for the years 1987 though 1995, in Table 8-6. These annual costs were calculated as the product of the per-vehicle cost to the consumer (from Table 8-5), the projected sales of LDDVs and LDDTs (from Table 8-4), and the diesel trap penetration rates (from Tables 1-4 and 1-7 in Chapter 1), using both the best estimate and "worst-case" sales projections.

III. Heavy-Duty Diesels

A. Introduction

This section is divided into subsections corresponding to those in the preceding discussion on light-duty diesels. First, the RPE of the manufacturing costs of trap-oxidizers, for different classes of HDDEs, are estimated. Second, the impact of these costs on sales and the capital investment requirements of the HDDE manufacturers are examined. Next, the increase in the total cost to the consumer of owning and operating an HDDE due to the particulate control regulations of the base case is estimated. Finally, the annual costs (for 1988 through 1995) of the base scenario are estimated. All of the costs presented in this section are in 1983 dollars. It is assumed that 1988 will be the first effective year of heavy-duty diesel particulate control regulations.

Much of the methodology used here was described in the section on light-duty diesels, therefore frequent references are made to the preceding material.

B. Trap-Oxidizer System Costs

1. Introduction and Assumptions

Although most heavy-duty emission regulations make reference to heavy-duty engines, which are generally tested separately on engine dynamometers in lieu of tests on the actual finished vehicles, for internal consistency this analysis refers to heavy-duty diesel vehicles (HDDVs).

As was shown previously (II.B.2.), the trap volume required for a given application can be related directly to the volumetric exhaust gas flow to be treated. The cost of both trap designs, catalyzed and non-catalyzed, was then shown to be a function of the required trap volume. A similar approach is used in this section for heavy-duty diesels.

Table 8-6

Annual Costs to the Nation of the Base Scenario for LDDVs and LDDTs (millions of 1983 dollars)

	Best Estimate Sales Projections		"Worst-Case" Sales Projections	
	LDDVs	LDDTs	LDDVs	LDDTs
Annual Cost:				
1987	52	16	95	21
1988	60	18	115	23
1989	66	20	131	26
1990	74	22	148	28
1991	76	23	156	31
1992	77	25	164	35
1993	78	26	172	39
1994	79	27	180	42
1995	80	29	187	46

The estimates of the RPE of manufacturing costs for light-duty diesels were based on the application of adjustment factors to the estimated manufacturing costs. Additional adjustment factors were included in the model developed by Lindgren[5] to compensate for inflation and production volume. With the exception of the adjustments for production volume, these factors are unchanged for the heavy-duty case.

In order to estimate standard average trap production levels for HDDVs, the number of different trap sizes required must be determined. In the Regulatory Analysis for the proposed heavy-duty diesel particulate control regulations,[14] the assumption was that four sizes of traps would be required to span the entire range of HDDVs. The grouping of HDDVs into size classes at that time, based on gross vehicle weight (GVW) classes, is shown below:

Group	GVW Classes
1	IIB*, III, IV
2	V, VI
3	VII
4	VIII

In this analysis, HDDVs are divided into only three groups, on the basis of both GVW classes and relative sales. Classes VII and VIII HDDVs are consolidated in one group, and Class V is placed in the same group as Classes IIB-IV. These groups are referred to in the rest of this section as medium-duty diesels (MDVs), light heavy-duty diesels (LHDVs), and heavy heavy-duty diesels (HHDVs), in order of increasing GVW. This is summarized below:

Group	GVW Classes	GVW (lbs.)
MDVs	IIB-V	8,501-19,500
LHDVs	VI	19,501-26,000
HHDVs	VII-VIII	26,001 and over

These groups have the advantage that each contains one of the three GVW classes that dominate HDDV sales (IIB, VI, VIII), while GVW classes having relatively low sales are grouped with them through similarity of application. This division is also consistent with the diesel manufacturers' typical grouping of HDDVs.[15,16]

The standard average production level, for traps for each of the three LDDV size classes, was estimated in the preceding section to be 200,000 annually. This figure was based on the best estimate diesel sales projections[3] and the projected rates of trap usage (Chapter 1). These projections also indicate that approximately half as many HDDVs will be sold, compared to sales of each LDDV size class, in each of the three groups defined above. The standard average production level for each HDDV group is then 100,000 annually.

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* Class IIB in this analysis refers to all vehicles in the traditional GVW Class II (6,001-10,000 lbs.) that EPA classifies as heavy-duty (GVW over 8,500 lbs., or frontal area over 45 square feet, or curb weight over 6,000 lbs.)

However, due to the large trap volume requirements for LHDVs and HHDVs, this analysis assumes that two traps (each with half the total volume required), will be fitted to those vehicles. The standard average trap production levels are then 100,000 for MDVs, and 200,000 each for LHDVs and HHDVs. Assuming the 12 percent learning curve used in the light-duty analysis, the adjustment factors for production volume are 1.136 for MDVs, and 1.0 (no adjustment) for LHDVs and HHDVs.

The trap volume requirements are calculated in the following two sections in the same way that the light-duty trap volume requirements were determined. Trap size is related to volumetric exhaust flow, which in turn is proportional to fuel consumption (inverse of fuel economy). This calculation requires estimates for the average new-vehicle fuel economy of each class (MDV, LHDV, HHDV) in the late 1980's and early 1990's. Actual projections of 1990 fuel economy for heavy-duty gasoline-powered vehicles (HDGVs)[15] were raised by 30 percent to account for the increased efficiency of diesel engines, giving the projections of 15.5 mpg (MDVs), 8.4 mpg (LHDVs), and 7.0 mpg (HHDVs) used in this analysis. These figures represent 1990 project average fuel economies for new heavy-duty vehicles. Thus, they are slightly higher than the fuel economy projections used in Chapter 2, which represent the entire heavy-duty diesel in-use fleet in 1990.

2. Non-Catalyzed (Corning) Trap

The trap volume requirements of non-catalyzed traps for heavy-duty applications are based, as in the light-duty case, on the successful testing of a 302 cubic inch Corning trap on a Mercedes-Benz 300D with fuel economy of 26 mpg. The trap volume requirements that result are 506 cubic inches for MDVs, 934 cubic inches for LHDVs, and 1,122 cubic inches for HHDVs.

As noted above, the magnitude of the trap volume requirements for LHDVs and HHDVs was high enough to assume that two traps will be fitted, per vehicle, with the total volume equal to the required size. The individual trap volumes are then 467 cubic inches for LHDVs and 561 cubic inches for HHDVs.

In section II.A.2., formulae were given that yielded the RPE of manufacturing cost as a function of trap volume. The equations 2A-2C, when adjusted for production level as discussed in the introduction to this section, become:

For MDVs:

$$RPE = \$26 + 0.358(V) \quad (4A)$$

For LHDVs and HHDVs:

$$RPE = \$23 + 0.318(V) \quad (4B)$$

Where:

V = volume of trap, in cubic inches.

Substitution of the trap volume requirements for V in equations 4A and 4B gives the RPEs of the heavy-duty traps. The 506 cubic inch traps for MDVs have an RPE of about \$207. For LHDVs, each of the 467 cubic inch traps needed has an estimated RPE of about \$172; the total RPE for two such traps (per-vehicle RPE) is \$343. The 561 cubic inch traps for HHDVs have an estimated RPE of about \$201 each, with a per-vehicle RPE for two such traps of \$403. All of these estimates, which are based on best estimate sales projections, are shown in Table 8-7.

3. Catalyzed (Johnson-Matthey) Trap

As in the light-duty case, calculation of catalyzed trap volume requirements is based on the successful testing of a 345 cubic inch trap on a Volkswagen Rabbit (fuel economy of 42 mpg). The projected fuel economy of various HDDVs was related to this fuel economy to obtain catalyzed trap sizes. The results are: 935 cubic inches for MDVs, 1,724 cubic inches for LHDVs, and 2,070 cubic inches for HHDVs. The assumption that the volume requirements for LHDVs and HHDVs would be met by fitting two traps of equal volume is also used here. The single-trap volumes are then 862 cubic inches for LHDVs and 1,035 cubic inches for HHDVs.

Equations 3A-3C were used in section II.A.3. to calculate the RPE of the manufacturing costs for light-duty catalyzed traps. When the adjustment factors for heavy-duty production levels are applied, the new equations are:

Table 8-7

Heavy-Duty Trap Costs (1983 dollars)

Best Estimate Sales Projections

Vehicle Class	Non-Catalyzed Trap	Catalyzed Trap
MDVs	$207	$636
LHDVs	$343	$1,051
HHDVs	$403	$1,252

Worst Case Sales Projections

Vehicle Class	Non-Catalyzed Trap	Catalyzed Trap
MDVs	$183	$560
LHDVs	$274	$1,007
HHDVs	$403	$1,252

For MDVs:

$$RPE = \$26 + 0.652(V) \tag{5A}$$

For LHDVs and HHDVs:

$$RPE = \$23 + 0.583(V) \tag{5B}$$

Where:

V = volume of trap, in cubic inches.

Therefore the MDV trap, with a volume of 935 cubic inches, has an estimated RPE of $636. Each of the 862 cubic inch traps for LHDVs has an estimated RPE of about $526, for a per-vehicle RPE of $1,051. The RPE of each 1,035 cubic inch HHDV trap is estimated to be about $626, or $1,253 for the two traps required. These estimates are all shown in Table 8-7.

All of the estimates for heavy-duty traps discussed above are based on the best estimate sales projections. Under the "worst-case" sales projections, sales of MDVs and LHDVs would double, with trap production for these vehicles also doubling. As was discussed in section II.B.1., the 12 percent learning curve assumed indicates that trap costs would be decreased 12 percent by a doubling of production. Since HHDV sales already represent nearly all sales in GVW Classes VII and VIII, they remain relatively constant under both sales projections. Thus, the "worst-case" sales projections have an insignificant effect on estimated HHDV trap costs. All of this information is summarized in Table 8-7.

4. Regeneration System Costs

Section I.C. described the components of both catalyzed and non-catalyzed trap regeneration systems. These basic systems will also be used for HDDV applications, with two changes that will have an impact on the cost estimates presented in Table 8-2. The use of two traps on LHDVs and HHDVs means that the quantities required of some regeneration system components will be doubled. In addition, the difference in the sizes of LDDV and HDDV engines will have an effect on the costs of the required stainless steel exhaust pipe, as discussed later.

Table 8-8 summarizes the estimated RPE of manufacturing cost for both catalyzed and non-catalyzed trap regeneration systems, for each of the three groups of HDDVs. The estimates shown include the doubled quantity of some of the components required for LHDV and HHDV applications.

Table 8-8

Heavy-Duty Regeneration System Costs (1983 dollars)

Hardware Item	Retail Price Equivalent		
	MDDV	LHDV	HHDV
Non-Catalyzed Trap:			
Burner Head	$7	$14	$14
Fuel Delivery System	$9	$18	$18
Ignition System*	$5-26	$10-31	$10-31
Auxiliary Combustion Air System	$30	$30	$30
Exhaust Diversion System*	$11-14	$15-18	$15-18
System Control:			
Temperture Sensors	$12	$24	$24
ECU	$37	$37	$37
Subtotal*	$111-135	$148-172	$148-172
Stainless Steel Exhaust Pipe**	$33	$53	$89
Total*	$143-168	$201-225	$237-261
Catalyzed Trap:			
Delayed In-Cylinder Fuel Injection Mechanism	$30	$30	$30
Auxiliary Combustion Air System (Reed Valve)	$6	$12	$12
System Control:			
Sensors	$12	$24	$24
ECU	$37	$37	$37
Subtotal	$85	$103	$103
Stainless Steel Exhaust Pipe**	$33	$53	$89
Total System Cost	$118	$156	$192

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* Explanations of ranges in costs are in section II.B.4.

** For exhaust pipes only, the assumed average production volume is 100,000 units.

In section II.B.4. it is shown that, of the current $75 cost of an electronic control unit (ECU),[2] about $10 is attributable to particulate control on a per-vehicle basis. Since ECUs are not projected to be in general use on heavy-duty vehicles before 1988, but will be required under the base scenario, a greater share of the total cost should be attributed to particulate control. This analysis assumes that the ECU will be applied solely for emission control purposes, and that it will be used for both particulate and NOx control. Thus, it's cost is divided equally between particulate and NOx control functions, yielding the ECU cost estimate for particulate control of $37 shown in Table 8-8.

The additional costs of a stainless steel exhaust pipe for light-duty diesels were estimated (II.B.4.) as about $16 for 4- and 6-cylinder engines, and about $27 for 8-cylinder engines. These costs presume a single exhaust manifold with both 4- and 6-cylinder engines, and a crossover system with the 8-cylinder engine. In the case of HDDVs, the majority of engine exhaust systems are the single, non-branching type. Fewer systems are of the dual exhaust type, where two entirely separate exhaust systems are used.[14] This analysis assumes that all HDDVs in GVW Classes IIB-V are manufactured with single exhaust systems, while for Class VI and larger HDDVs, 75 percent are manufactured with single exhaust systems and 25 percent with dual exhaust systems.[14]

In the Draft Regulatory Analysis to the Heavy-Duty Diesel Particulate NPRM,[14] it was stated that the basic design of the LDDV 6-cylinder engine exhaust pipe should be the best analogue of the exhaust pipe design of HDDVs with single exhaust systems. This is also assumed in these estimates. For HDDVs with dual exhaust systems, the resulting cost estimates are doubled (i.e., two stainless steel exhaust pipes are assumed to be used).

The estimated cost of converting from a standard steel to a stainless steel exhaust pipe for an LDDV with a 6-cylinder engine was given (II.B.4.) as $16, which included credit for the deleted standard steel pipe. The corresponding costs for HDDVs are calculated by assuming a direct relationship of material cost to engine displacement. The typical LDDV 6-cylinder engine is assumed to have displacement of 3.7L. The typical engine displacements for heavy-duty diesels are assumed to be 6.2L (MDVs), 8.2L (LHDVs), and 13.9L (HHDVs). For GVW Classes VI, VII, and VIII vehicles (LHDVs and HHDVs), the average per-vehicle cost is calculated by assuming that 75 percent of these vehicles will require one pipe and 25 percent of them will require two, as discussed above.

The result of these calculations is the estimated per-vehicle average RPE of manufacturing cost for stainless steel exhaust pipes: $33 for MDVs, $53 for LHDVs, and $89 for HHDVs.

These costs are also shown in Table 8-8. Except for the exhaust pipes and the doubled quantities of some components, the cost estimates in Table 8-2 for light-duty trap regeneration systems remain unchanged in Table 8-8.

5. Total Trap-Oxidizer System Costs

The sum of the estimated costs of the trap and corresponding regeneration system is the estimated total trap-oxidizer system cost. These sums are shown, for both trap types and under both sets of sales projections, in Table 8-9.

As was the case in the light-duty analysis, the cost ranges are small relative to the absolute costs. Thus, only the midpoints of these ranges are used in Table 8-9 and in the remaining analysis. The non-catalyzed trap, which is much less expensive and appears to be the preferred design of heavy-duty diesel manufacturers, is the basis of the rest of the analysis.

C. Economic Impact on Manufacturers

In this section, the impact of the base particulate control scenario on manufacturers' heavy-duty diesel sales, capital investments, and cash flow are estimated. As for light-duty, only the costs of the trap-oxidizer system are considered. Certification costs have already been shown to be negligible.[14]

1. Impact on Manufacturers' Sales

Estimating the impact of the base scenario on HDD sales is considerably more difficult than was the case for light duty. There are two main reasons for this. First, little research has been conducted into the economic elasticities at work in the heavy-duty diesel market, and relevant data are scarce. In addition, there are complicating factors such as the division of heavy-duty diesels into three groups (MDV, LHDV, HHDV), and the relatively insignificant sales of vehicles in some GVW Classes (III, IV, and V). Thus the analysis and estimates presented in this subsection must be considered to be, at best, rough approximations.

The discussion and the estimated impact of the base scenario on sales in each HDD group presented in this section are based primarily on a report recently prepared for EPA by

Table 8-9

Total Heavy-Duty Trap-Oxidizer System Costs (1983 Dollars)

Best Estimate Sales Projections

Vehicle Class	Non-Catalyzed Trap	Catalyzed Trap
MDVs	$363	$754
LHDVs	$556	$1,207
HHDVs	$652	$1,444

Worst Case Sales Projections

Vehicle Class	Non-Catalyzed Trap	Catalyzed Trap
MDVs	$339	$678
LHDVs	$487	$1,163
HHDVs	$652	$1,444

Jack Faucett Associates (JFA).[17] JFA conducted a thorough literature search and surveyed a number of knowledgeable individuals, including members of the heavy-duty vehicle and engine industries, in order to develop the economic elasticity estimates used here.

Two kinds of price elasticity, own-price and cross-price, must be considered. Own-price elasticity refers to the change in the demand for vehicles of a given category resulting from a change in the purchase price of vehicles in that same category. Cross-price elasticity takes into account the shifts that may occur, from diesel to gasoline-fueled engines or conversely, as a result of changes in the purchase price of vehicles of one or both engine types within a given category.

In the heavy-duty market, distinct own-price elasticities exist for each engine type (diesel or gasoline fueled), within each GVW class (IIB through VIII). JFA supplied estimates of own-price elasticity for HDDs in Classes IIB, VI, VII, and VIII; no estimates were given for Classes III, IV, and V due to low sales.[17] These estimates are applied to the three groups under consideration here by assuming that own-price elasticity for MDVs is approximately equal to that of Class IIB alone, due to the very low sales of vehicles in the other GVW classes. HHDV own-price elasticity is approximated by the sales-weighted average of the elasticities of Classes VII and VIII. The estimates for Class VI are also the estimates for LHDVs, by definition of LHDV.

The best estimate and "worst-case" sales projections, for each of the three HDD groups, for 1990 and 1995 are shown in Table 8-10. Only the sales projections under the relaxed regulatory scenario are given. Since there is considerable uncertainty associated with the elasticity estimates used, the impact on sales of the base regulatory scenario are given in Table 8-10 as percent reductions from relaxed scenario sales.

Cross-price elasticity is a directional concept, depending on whether "from diesel to gasoline fueled" or "from gasoline fueled to diesel" is being considered. In this analysis only the former is of interest: Given an increase in the purchase price of HDDs in a given category, the own-price elasticity estimates how many sales are lost in that category, and the cross-price elasticity estimates how many of those "lost" sales are compensated for by increased sales of gasoline-fueled engines in the same category.

The uncertainties in the cross-price elasticity estimates used are fairly substantial. Although not shown in Table 8-10, the results of using the estimated cross-price elasticities are discussed below.

Table 8-10

Heavy-Duty Diesel Sales Projections (in thousands)

Best Estimate Sales Projections

Vehicle Class	1990	1995**	Reduction Due to Base Scenario***
MDVs	124	170	4.8%
LHDVs	94	126	2.2%
HHDVs	159	186	1.0%

Worst Case Sales Projections

Vehicle Class	1990	1995**	Reduction Due to Base Scenario***
MDVs	248	340	4.8%
LHDVs	188	252	2.2%
HHDVs	159	186	1.0%

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* California sales included.

** These sales figures are extrapolated from EPA sales projections for 1985 and 1990.

*** Percent reduction in relaxed scenario sales, applicable to both 1990 and 1995 projections.

Of the 4.8 percent reduction in MDV sales projected to occur under the base control scenario, over a third are estimated to be made up by increased sales of gasoline-fueled engines in Classes IIB-V. Thus, the net reduction in sales of all engines in Classes IIB-V is estimated to be approximately 3 percent. Similarly, the net reduction in LHDV sales is estimated to be approximately 2 percent. For HHDVs, a drop in sales of about 1 percent is projected to occur under the base scenario, and only about one in 50 of those "lost" sales is projected to be offset by new gasoline-fueled engine sales in Classes VII and VIII.

It should also be noted that the own-price and cross-price elasticities estimated by JFA were based only on changes in the initial purchase price. The effects of increases in operating and maintenance (O&M) costs are more difficult to incorporate into the model. In this analysis, the increase in O&M costs (net present value in year of vehicle purchase, 10 percent discount rate) was considered to be part of the initial purchase price increase. Although this is not appropriate, strictly speaking, it is an adequate approximation when the uncertainties inherent in the elasticity estimates are taken into account.

2. Capital Investment and Cash Flow Effects

Implementing a trap-based particulate standard for heavy-duty diesels should have only minor effects on the capital expenditures of HDDV manufacturers. The reasons are basically the same as discussed for light-duty in section II.B., and are briefly recapped below.

It is quite unlikely that any heavy-duty manufacturer will choose to make the necessary investments for the production of trap-oxidizers, as the sophisticated technology required has already been developed by other firms. In addition, the production volumes of most individual manufacturers will be far too small to justify establishment of in-house trap-production capability. Thus for heavy-duty as well as light-duty, the bulk of the investments required for trap-oxidizer production will be financed by emission control equipment manufacturers. Future R&D investments by the manufacturers are difficult to estimate, but should not be so high as to adversely affect other investment plans.

The cash flow impact of these regulations is limited to the inventory of traps, individually and on partially manufactured HDDVs. This investment is recovered upon sale of the trap-equipped HDDV, and the sales turnover period of HDDVs is short (generally less than four months). The short inventory period, and the relatively small amount of cash represented, should not significantly affect the cash flow of any manufacturer.

D. Total Cost to the Consumer

The total cost to the consumer is the sum of the costs of the trap-oxidizer system, shown in Table 8-9, and the costs of the 2 percent fuel-economy penalty[4] and increased maintenance costs, less any maintenance savings. As in the light-duty analysis, it is assumed that manufacturers pass all of their costs increases through to the retail purchaser.

The costs of the 2 percent fuel-economy penalty are estimated by the same methods used for light-duty diesels in section II.D. The information used in this calculation is: $1.20 per gallon average diesel fuel cost; 10 percent discount rate; new-vehicle fuel-economy estimates of 15.5 mpg (MDVs), 8.4 mpg (LHDVs), and 7.0 mpg (HHDVs); annual average VMT of 12,000 (MDVs), 20,000 (LHDVs), and 47,500 (HHDVs); and lifetime average VMT of 120,000 (MDVs), 200,000 (LHDVs), and 475,000 (HHDVs). When this information is used as described earlier, the net present value of the lifetime fuel-economy penalty, in the year of vehicle purchase, is $126 (MDVs), $386 (LHDVs), and $917 (HHDVs). This is summarized in Table 8-11.

The trap should be maintenance-free, but the regeneration system will require maintenance once during the lifetime of the HDDV, after approximately five years of operation. For light-duty diesels, the discounted cost of regeneration system maintenance is estimated at $22 (Table 8-5). This cost should be applicable without adjustment to MDVs, which will be equipped with a single trap. For LHDVs and HHDVs, with two traps per vehicle, this estimate is simply doubled to $44. Table 8-11 also shows these estimates.

A maintenance savings will result from the use of stainless steel exhaust pipes, which eliminate the need for periodic replacement of standard steel exhaust pipes. On average, the total per-vehicle savings would range from $39 (MDVs) to $97 (HHDVs) over the vehicle lifetime, using a 10 percent discount rate and an appropriate schedule for HDDV standard steel exhaust pipe replacement.[14]

The components of total consumer cost discussed, as well as the totals, are shown in Table 8-11. The total consumer costs are given for both best estimate and "worst-case" sales projections. Also in Table 8-11 is the estimated overall cost of owning and operating an HHDV over its lifetime, in terms of net present value in year of purchase (1983 dollars).[18] As can be seen, the impact of particulate control on this overall cost is small, about 0.6 percent.

Table 8-11

Total Cost to Consumers of Owning and Operating a Heavy-Duty Diesel Equipped with a Trap-Oxidizer (1983 dollars)*

	MDDV	LHDV	HHDV
Trap-Oxidizer System:			
Best Estimate Sales Projections	$363	$556	$652
"Worst-Case" Sales Projections	339	$487	$652
Fuel Economy Penalty	$126	$386	$917
Maintenance Costs	$22	$44	$44
Maintenance Savings	($39)	($61)	($97)
Total:			
Best Estimate Sales Projections	$472	$925	$1,516
"Worst-Case" Sales Projections	$448	$856	$1,516
Total Cost of Owning and Operating a HHDV[16]	-	-	$274,911
Cost Increase Due to Trap-Oxidizer	-	-	0.6%

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* All costs are discounted to the year of vehicle purchase using a 10 percent rate.

E. Annual Costs

The annual costs of the base regulatory scenario are shown, for the years 1988 to 1995, in Table 8-12. These costs were calculated by multiplying the net present value of the total cost to the consumer, per vehicle, by annual sales.

The costs summarized in Table 8-12 are shown for two possible situations: trap-oxidizers are applied to all HDDVs, and to only 70 percent of HDDVs. As was discussed in Chapter 1, the lower trap usage rate would be adequate if emissions averaging is made available to HDDV manufacturers and 85 percent efficiency ceramic traps are used on all trap-equipped vehicles.

Table 8-12

Annual Costs to the Nation of the Base Scenario for Heavy-Duty Diesels (millions of 1983 dollars)

	Best Estimate Sales Projections		"Worst-Case" Sales Projections	
	Trap Usage		Trap Usage	
	70%	100%	70%	100%
Annual Cost:				
1988	239	341	325	464
1989	251	358	344	491
1990	262	375	362	517
1991	295	422	402	574
1992	328	469	443	633
1993	361	516	480	686
1994	395	564	521	744
1995	430	614	564	805

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CHAPTER 9

COST EFFECTIVENESS

I. Introduction

Cost effectiveness is a relative measure of the economic efficiency of taking an action to achieve a specified goal. It is primarily useful in comparing alternative means of achieving that goal. In the context of this study, the goal is to reduce particulate emissions, or perhaps more importantly, to reduce ambient levels of particulate where people are exposed. In this case, cost effectiveness is expressed in terms of the dollar cost per ton of particulate emission controlled.

The primary purpose of this chapter is to determine the cost effectiveness of the base scenario for each diesel vehicle subgroup so that comparisons among these subgroups can be made, and so that these mobile source strategies can be compared on a relative basis to non-mobile source strategies. The baseline or starting point for evaluating the cost effectiveness of the base scenario is the relaxed scenario, which itself provides some level of control. The relaxed scenario was chosen as the baseline instead of a totally uncontrolled case for two principal reasons.

First, the 0.60 g/mi standards for light-duty diesel vehicles and trucks (LDDs) have already been implemented. For heavy-duty diesel vehicles (HDDVs), compliance with the 0.60 g/BHP-hr standard should not be difficult and would likely preceed a trap-based standard. Because of this, it is most appropriate to evaluate the base scenario against the baseline which exists at the time these new requirements become effective, i.e., the relaxed scenario.

Second, the relaxed scenario itself represents a very small degree of control. Almost all LDDs are already emitting at or below the standards. Fleetwide, most HDDs are emitting very near the standard. Therefore, there is little practical difference between the relaxed scenario and a totally uncontrolled case, although use of the relaxed scenario will make the cost effectiveness of the base scenario slightly worse than if the uncontrolled case were used. Hence, for both of these reasons, it is appropriate to evaluate the base scenario as being incremental to the relaxed scenario. (The base and relaxed scenarios are described fully in the Introduction.)

To determine cost effectiveness, two pieces of information are necessary: the costs and emission reductions of the strategies to be examined. The measure of cost will be the annualized net present value of all purchase, operating, and maintenance costs. Emission reductions will be determined on an annual basis in terms of either total, inhalable or fine particulate.* The three classes of suspended particulate as examined in order to focus the analysis on the most important particulate matter with respect to public health and welfare. As determined in Chapter 6, fine and inhalable particulate have the primary effect on human health. As determined in Chapter 4, only fine particulate affects visibility. As outlined in Chapter 7, all particulate can participate in soiling.

The remainder of this analysis is divided into three major sections. The first section estimates the cost-effectiveness ($/metric ton) for the base control scenario (relative to the relaxed scenario) and concludes with a comparison of these figures for the various diesel subgroups on a nationwide and urban basis. The second section of the analysis will estimate cost-effectiveness values for several stationary sources. The third section will conclude the analysis by: 1) applying a discount factor to the cost-effectiveness values for both mobile and stationary sources to account for their relative air quality impacts, and 2) comparing the cost-effectiveness of diesel particulate control to those of stationary sources.

It should be noted that while cost-effectiveness analysis is valuable in discerning relative economic efficiencies of various actions to achieve a specific goal, it cannot provide definitive information on the point at which the costs of controlling any particular source exceed the economic benefits (i.e., dollar value) associated with control. This latter type of evaluation can be made only by using cost-benefit analysis, which is beyond the scope of this report. A cost-benefit study of mobile source diesel particulate standards has been recently completed for EPA under contract[1] and the reader is referred to that study and others in the literature for more information on the benefits of controlling diesel particulate.

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* Total particulate is all suspended particulate matter regardless of diameter, inhalable particulate is considered to be all particulate matter less than 10 micrometers in diameter, and fine particulate is considered to be all particulate matter less than 2.5 micrometers in diameter.

II. Cost Effectiveness of Controlling Particulate Emissions from Diesel Vehicles

A. Methodology

In this section, the cost effectiveness of proceeding from the relaxed to the base scenario for five diesel vehicle subgroups will be estimated and compared. These subgroups include light-duty diesel vehicles (LDDVs), light-duty diesel trucks (LDDTs), and three subgroups of HDDVs: medium-duty vehicles (MDVs), light heavy-duty vehicles (LHDVs), and heavy heavy-duty vehicles (HHDVs).

Most previous EPA cost-effectiveness analyses for mobile source emissions have determined cost effectiveness using total lifetime costs discounted to the year of vehicle purchase and undiscounted lifetime benefits. However, this approach is somewhat simplistic, since it disregards the fact that the emission reductions cannot be obtained at the time of vehicle purchase, when the cost of control is determined. Because of this, the cost-effectiveness value calculated is entirely dependent on the point in time costs are determined, which is somewhat arbitrary.

It would be more appropriate if costs could be allocated to each period of time in which benefits were produced and in proportion to the size of these benefits. The result would be a cost effectiveness which is applicable at any point in that life as well as over the entire life of the vehicle.

This can be done here for mobile sources through the use of two simplifying assumptions which will not affect the accuracy of the cost-effectiveness comparisons. First, it will be assumed that the number of miles a diesel is driven annually is constant throughout its useful life. This simplifies the determination of the miles producing emission reductions each year. Second, the per-mile emission reduction occurring at the vehicle's half life will be assumed to apply throughout its life. This assumption allows direct use of the emission results of Chapters 1 and 2, since the analysis there also assumed that emissions were constant with mileage except for the effect of trap failure. This only results in a slight underestimation of emission reductions early in life, with a compensating overestimation late in life. The overall effect on cost effectiveness is negligible.

With the use of these two assumptions, the annual emission reduction throughout a vehicle's life becomes constant and the cost of control can simply be allocated equally (using discount theory) to each year of the vehicle's life. This latter annualized cost is simply an annuity equivalent to the total cost of control discounted to the year of vehicle purchase, which was determined in Chapter 7. Costs will be addressed first and then emission reductions, followed by calculation of the cost-effectiveness values.

B. Costs of Control

Essentially all of the cost information necessary for the cost-effectiveness calculations has been developed in Chapter 8. Tables 9-5 and 9-11 of that chapter contain detailed cost information on the purchase and operating cost impacts for LDDVs, LDDTs, and HDDVs. These costs are given in 1983 dollars, discounted at 10 percent to the year of vehicle purchase.

This cost-effectiveness analysis does not require the level of disaggregation given in Table 9-5 for LDDVs and LDDTs (e.g., small, medium, and large LDDVs as opposed to simply LDDVs). Therefore, the costs presented will be combined to obtain total lifetime consumer costs for LDDVs and LDDTs. As outlined in Chapters 1 and 8, the largest vehicles are likely to be equipped with trap-oxidizers first, since they are the highest emitters. Since the trap usage rates under the base scenario (22 percent for LDDVs and 9 percent for LDDTs) are below the projected sales fractions of large LDDVs (26 percent) and full-size LDDTs (66 percent),[2] only the largest size vehicles in each class are likely to have traps. Thus, the lifetime costs for these largest vehicles will be used here.

Table 9-1 shows the discounted total lifetime consumer costs for each of the five diesel vehicle groups. (HHDV costs can be take directly from Chapter 8.) Only those costs for the best estimate sales scenarios are shown. Costs for worst case sales would be 0-4 percent lower, because of economies of scale. (Each vehicle class has a different factor since the relationship between best estimate and worst case sales is different for each vehicle class.)

These discounted total costs can be annualized (at mid-year) over the appropriate average lifetime for each of the diesel vehicle classes using present value theory. The expected vehicle lifetimes and the resultant annualized costs are shown in Table 9-1.

For LDDVs and LDDTs, trap-oxidizers will be used only on the portion of each manufacturer's sales necessary to bring the manufacturer's sales-weighted particulate levels down to the required standard. Since the particulate reduction benefits will be measured on a fleetwide basis, but costs shown in Table

9-1 only apply to a portion of the fleet, these costs must be spread over the entire fleet. This can be accomplished by multiplying the annualized costs of Table 9-1 by the percent of vehicles requiring traps (taken from Tables 1-4 and 1-7 of Chapter 1). Since the base scenario does not assume the availability of an averaging concept for HDDVs, this affects only LDDVs and LDDTs. (Without averaging, all HDDVs will use trap-oxidizers and no adjustment needs to be made.) These fleet-average annualized costs for LDDVs and LDDTs are also shown in Table 9-1. With averaging, about 70 percent of all HDDVs would require traps and the fleetwide costs shown in Table 9-1 would be reduced by approximately 30 percent.

C. Diesel Particulate Emission Reductions

Calculation of the annual diesel particulate emisson reduction accompanying the base scenario requires information on annual vehicle miles travelled (VMT) and the emission rates under the two control scenarios. Table 9-2 shows the average annual mileage for each of the five diesel vehicle subgroups, which were derived from each subgroup's average lifetime mileage and average life (also shown).

Vehicle particulate emission rates (g/mi) tend to increase gradually with mileage, in a manner in which can be characterized as linear over the life of the vehicle. Thus, for either the relaxed or base scenario, one can conceptualize a stream of annual particulate emissions, increasing by a constant amount each year. If the emissions in each year for the base scenario were subtracted from the emissions in each year for the relaxed scenario, a stream of emission reductions would be created. Costs could then be allocated to this stream of benefits to provide a constant and applicable cost effectiveness throughout the vehicle's life.

As already mentioned in the previous section, a close approximation to this can be obtained by ignoring the small change in emissions with time and determining the emission reduction at the vehicle's half life. The half life for LDDVs and HDDVs is approximately the fifth year; for LDDTs it is the sixth year.

Half-life emission rates for the five diesel vehicle subgroups were taken from Chapter 2, and are shown in Table 9-2. Unlike LDDs, the half-life emission rates for HDDVs differ somewhat for each model year under both scenarios because the fuel efficiencies and, hence, emission characteristics of these vehicles are expected to improve in future years. To simplify the cost-effectiveness computations, half-life emission rates for 1990 model year HDDVs are used.

Table 9-1

Base Scenario Costs (1983 dollars)*

	LDDV	LDDT	MDV	LHDV	HHDV
Lifetime Costs for Base Scenario	$266	$252	$472	$925	$1,516
Vehicle Lifetime	10 yrs	11 yrs	8 yrs	11.5 yrs	10.5 yrs
Annualized Cost For A Trap-Equipped Vehicle	$41	$37	$84	$132	$228
Percent of Vehicles With Trap-Oxidizers	22.3	7.6	100	100	100
Fleet Average Annualized Cost Per Vehicle	$9.20	$2.80	$84	$132	$228

+ Discounted at 10 percent to year of vehicle purchase, best estimate sales.

Table 9-2

Average Annual Vehicle Miles of Travel

	LDDV	LDDT	MDDV	LHDDV	HHDDV
Average Lifetime Mileage	100,000	120,000	110,000	268,000	529,000
Lifetime (years)	10	11	8	11.5	10.5
Average Annual Mileage	10,000	10,900	13,800	23,300	50,100
Vehicle Emission Rates at Half Life (g/mile)					
Relaxed Scenario	.270	.280	.818	1.381	2.151
Base Scenario	.204	.261	.381	.642	1.034
Difference	.066	.019	.437	.734	1.117
Annual Emission Reduction (grams)	660	210	6,030	17,100	55,960
(metric tons)	6.6×10^{-4}	2.1×10^{-4}	$6.03 \times 10^{+3}$	0.0171	0.0560

Also, to further simplify the analysis, a single emission rate is used to represent Classes IIb and III-V as MDDVs. This was determined by sales-weighting the emission rates for the two classes using projections for the 1990 model year from Reference 3. These simplifications have no significant effect on the results of this analysis. It is also worth noting that, for the base scenario, where trap-oxidizers are used on all vehicle subgroups to gain the emission reductions, the emission rates include the effect of trap-oxidizer failures.

The annual emission reductions for each diesel vehicle subgroup can now be calculated by simply finding the difference in the emission rates from the relaxed and base scenarios and multiplying by the average annual mileage. These are shown in Table 9-2 in both grams and metric tons of diesel particulate controlled.

Diesel particulate matter is very small in size, with mass mean diameters varying from 0.05 to 0.2 micrometers. As such, essentially all diesel particulate falls into the fine particulate category.[4,5,6] Therefore, the emission reductions for total particulate given in Table 9-2 also represent the emission reductions for inhalable and fine particulate.

D. Cost-Effectiveness Values for Diesel Vehicles

The cost effectiveness of the base scenario is computed by dividing the fleet average annualized costs from Table 9-1 by the annual emission reductions from Table 9-2. The resulting cost-effectiveness values for the five diesel classes are given in Table 9-3 in the form of 1983 dollars per metric ton. The cost effectiveness of diesel particulate control is essentially equivalent for LDDVs and LDDTs (at $13,000-14,000 per metric ton), but appears to be better for MDVs and especially LHDVs and HHDVs.

These cost-effectiveness values presume the availability of averaging for LDDVs and LDDTs, but not for MDVs, LHDVs, or HHDVs. In Chapter 8, it was determined that HDDV compliance costs would drop approximately 30 percent if an averaging approach was used. Revised values incorporating averaging for HDDVs are also shown in Table 9-3. As can be seen, this change makes the control of HDDVs even more attractive relative to that of LDDVs or LDDTs.

It is important to note that these cost-effectiveness values consider all emission reductions, regardless of whether the reduction occurs in an urban or rural area. Since the great majority of Americans exposed to violations of the NAAQS

Table 9-3

Cost-Effectiveness Values
Total, Inhalable, and Fine Diesel Particulate*

	LDDV	LDDT	MDV	LHDV	HHDV
Average Annualized Cost ($)	\$9.20	\$2.80	\$84	\$132	\$228
Annual Emission 0.0560 Reduction (metric tons)	6.6×10^{-4}	2.1×10^{-4}	6.03×10^{-3}		0.0171
Cost Effectiveness (\$/metric ton)	13,900	13,300	13,930	7,740	4,070
Cost Effectiveness with Averaging for HDDVs (\$/metric ton)	13,900	13,300	9,750	5,420	2,850
Urban Cost Effectiveness with Averaging for HDDVs (\$/metric ton)	23,400	27,200	20,000	11,100	10,600

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* Cost-effectiveness values are the same for total, inhalable and fine particulate.

for particulate matter live in urban areas and since diesel particulate concentrations are greatest in these locations, the control of diesel particulate in urban areas should receive the greatest emphasis. This is not to suggest that the benefits associated with controlling diesel particulate in rural areas are unimportant, however. In fact, diesel particulates emitted in rural areas, or transmitted through the atmosphere into isolated regions, may adversely affect agriculture, visibility, etc. Nonetheless, because urban areas account for the greatest population exposed to NAAQS violations, control of these emissions is most important from a public health perspective. Hence, evaluating diesel particulate control strategies on an urban basis is desirable.

As estimated in Chapter 2, the five diesel vehicle subgroups accumulate different fractions of their annual VMT in urban areas: LDDVs, 59.4 percent; LDDTs, MDVs, and LHDVs, 48.8 percent; and HHDVs, 26.9 percent. Urban cost-effectiveness values taking these fractions into account are also shown in Table 9-3 with averaging for all classes. A comparison of these values shows all five figures to be much more similar than before; however, the control of HDDVs still appears to be more cost effective than that of LDDVs and LDDTs.

These urban cost-effectiveness values have been developed only for comparison among the five diesel subgroups. The nationwide cost-effectiveness values of Table 9-3 will be used in comparisons with stationary source controls for two reasons. First, urban cost-effectiveness values are not available for stationary sources. Second, the use of urban values for only mobile sources would artifically make the cost effectiveness of diesel particulate controls appear worse relative to that of stationary sources controls.

III. Cost Effectiveness of Controlling Particulate Emissions from Selected Stationary Sources

A. Introduction

One means of gauging the appropriateness of controlling diesel particulate emissions is to compare the cost effectiveness of diesel particulate control against the cost effectiveness of controlling particulate emissions from stationary sources. This section of the analysis will be devoted to developing cost-effectiveness values for stationary sources. The following section will then develop a methodology for converting the cost-effectiveness values derived both here and in the previous section into values which are comparable on an air qualtiy basis. These latter values will be used in the comparison of mobile and stationary source controls.

A total of eight stationary sources have been selected for study, based on the availability of control cost information and emission reductions on a total, inhalable, and fine particulate basis. These eight sources are listed below:

Source	Particulate Control System
Borax Fusing Furnace	Venturi Scrubber
Wet Cement Kiln	Electrostatic Precipitator
Medium-Sized Industrial Boiler	Baghouse
Electric Utility Coal-Fired Generator	Electrostatic Precipitator
Kraft Recovery Furnace	Electrostatic Precipitator
Kraft Smelt Tank	Venturi Scrubber
Rotary Lime Kiln	Electrostatic Precipitator and Baghouse
Electric Arc Furnace (steel)	Baghouse

Two sets of data and, therefore, two different approaches will be used in this analysis. Costs and emission reductions for the first two sources listed above will be developed here from data contained in a recently published EPA report on control techniques for stationary source particulate emissions (herein after referred to as the Control Techniques document).[7] Cost-effectiveness values for the last six sources listed above have already been developed in a previous EPA analysis.[8] These will be used directly here, with some adjustments to the costs due to inflation, and, where data permits, some adjustment to the amount of the inhalable particulate benefits due to a change in the assumed maximum diameter for inhalable particulate from 15 to 10 micrometers.

The particle size distributions, source and emission control systems characteristics, and costs used in this analysis are based on the best available data and are representative of the sources being considered. However, it is important to note that all of the values used would likely vary from source to source within each source category, so this data and the analysis which follows cannot be routinely applied to every individual source. Stationary source emission control systems are not standardized, but are designed to meet the needs of each user. However, even with these qualifiers, the cost-effectiveness values developed here will serve as a valid basis of comparison with the cost-effectiveness of diesel particulate control.

B. Cost Effectiveness of Controlling a Borax Fusing Furnace and a Wet Cement Kiln

1. Costs of Control

Given the necessary information on source and emission control system characteristics, Volume 1 of the Control Techniques document mentioned above contains a number of correlations which can be used to estimate the annualized costs of particulate emission control systems. These annualized costs include both capital, direct and indirect operating costs, and have been developed from data presented in a more detailed EPA report.[9]

The annualized costs given in the Control Techniques document cover 8,700 hours per year of operation, or essentially continuous use. This is probably unrealistic since a normal downtime for scheduled and unscheduled maintenance of approximately 10 percent would be expected. Using 8,700 hours per year without downtime will tend to improve cost effectiveness, since fixed costs remain during downtime but the emission reduction is completely lost. However, since no accurate estimates of downtime experienced by the various stationary sources are available, no adjustment will be made here. (Assuming continuous operation happens to also be consistent with the manner in which the cost-effectiveness values were calculated in the draft HDD particulate regulatory analysis, which are addressed in Section C.)

The Control Techniques document presents costs in January 1980 dollars. Updating them to 1983 dollars using the producer price index for all industrial commodities[10] leads to an annualized cost increase of about 32 percent.

Table 9-4 presents the annualized costs for the borax fusing furnace, the wet cement kiln, and the values of the particulate control system parameters used to estimate these costs from the previously mentioned figures. In some cases, the values for these parameters were taken from the Control Techniques document. In other cases, the values were based on emissions data in EPA's Office of Air Quality Planning and Standards.[11]

2. Emission Reductions

As was done for diesels, emission reductions for stationary sources will be developed on a total, inhalable, and fine particulate basis. Table 9-5 presents size-specific emissions data for the uncontrolled and controlled cases for each source. The first column shows particulate concentration

Table 9-4

Parameter Values and Annualized Control Costs for Selected Stationary Source Particulate Controls (1983 dollars)

Source	Control System	Control System Parameters	Exhaust Gas Rate (Am^3/sec)	Annualized Cost
Borax Furnace	Scrubber	Delta = 11 kPa	38	$1,170,000
Wet Cement Kiln	ESP	SCA =120 m^2/(m^3/sec)	130	$1,320,000

Table 9-5

Emissions Data for Borax Fusing Furnace and Wet Cement Kiln

	Particulate Size Basis					
	Total		Inhalable		Fine	
	Mass Concentration (mg/DNCM)	Annual Emissions (metric tons)	Mass Concentration (mg/DNCM)	Annual Emissions (metric tons)	Mass Concentration (mg/DNCM)	Annual Emissions (metric tons)
Borax Fusing Furnace						
Uncontrolled	784	786	596	598	531	532
Controlled	24.3	24	20.6	20	19	20
Reduction	--	762	--	578	--	512
Wet Cement Kiln						
Uncontrolled	2.02×10^7	3.29×10^7	14,800	24,000	6,000	9,770
Controlled	67.4	110	109	109	63.1	103
Reduction	--	3.29×10^7	--	23,991	--	9,667

in terms of milligrams per dry nominal cubic meter (mg/DNCM) of exhaust gas. The annual emission levels were determined by multiplying the mass concentrations by the exhaust gas flow rates expressed in dry nominal cubic meters. These exhaust gas flow rates were estimated to be 32 DNCM/sec for the borax fusing furnace and 52 DNCM/sec for the wet cement kiln using the actual exhaust gas flow rates from Table 9-4 and the appropriate adjustment factors for temperature, pressure, and moisture content.

Now, given the exhaust gas flow rate in DNCM/sec, size specific mass concentration in mg/DNCM before and after control, and an annual operation period of 8,700 hours per year, the annual metric tons of particulate emissions and reductions by particle size can be calculated. Table 9-5 shows these annual emission rates on a particle size basis before and after control for both the borax fusing furnace and wet cement kiln, assuming a constant reduction efficiency with time. Subtracting emission rates before and after control gives the emission reduction.

Given the annualized cost values in Table 9-4 and the annual emission reduction in Table 9-5, cost-effectiveness values on a total, inhalable, and fine particulate basis can be determined. These are shown in Table 9-6.

C. Update of Previously Developed Cost-Effectiveness Values

In previous analyses, EPA developed cost-effectiveness values for a number of different stationary sources and particle sizes.[8] These values require two adjustments before being used in this analysis. First, costs must be updated from 1980 to 1983 dollars. This can be accomplished using the 32 percent change in the producer price index for all industrial commodities which was also used above.

Second, the inhalable particulate emission reductions estimated previously also require some adjustment due to a change in the assumed cutoff diameter from 15 micrometers in the 1980 analysis to 10 micrometers in the present analysis. This reduction in emission benefits will in turn lead to an increase in the relative cost effectiveness on an inhalable particulate basis.

After reviewing the sources for the original estimates and other data developed since that time, entirely new estimates for the mass percent of inhalable particulates have been developed for the electric utility and the electric arc furnace. The inhalable fraction of electric utility

Table 9-6

Cost Effectiveness for Stationary Sources[1,2]
(1983 Dollars per metric ton)

Source	Particulate Size Basis		
	Total	Inhalable	Fine
Wet Cement Kiln	[8]	55	136
Kraft Smelt Tank	250	299	455
Electric Arc Furnace[3]	924	1,440	1,452
Electric Utility[4]	1,254	1,805	4,092
Industrial Boiler[5]	1,320	1,848	5,544
Rotary Lime Kiln (ESP)[6]	1,584	1,980	3,168
Borax Fusing Furance	1,532	2,021	2,281
Rotary Lime Kiln (Baghouse)[6]	1,716	2,112	3,300
Kraft Recovery Furnace[7]	1,678	2,145	3,055

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[1] Ranked according to Inhalable Particulate Cost Effectiveness.

[2] For simplification, the midpoint of the ranges were used where applicable.

[3] Direct evacuation with 90 percent efficient canopy hood versus direct evacuation with open roof.

[4] High efficiency ESP (0.03 lb/10^6 BTU) versus lower efficiency ESP (0.1 lb/10^6 BTU).

[5] Baghouse (0.03 lb/10^6 BTU) versus cyclone (0.3 lb/10^6 BTU).

[6] High efficiency ESP (0.6 lb/ton limestone) versus lower efficiency ESP (0.6 lb/ton limestone) for 500 TPD plant; baghouse (0.3 lb/ton) versus lower efficiency ESP for 125 TPD plant.

[7] High efficiency ESP (99.5 percent) versus lower efficiency ESP (99.0 percent).

[8] Less than $1 per metric ton.

particulate was decreased from the 90-100 percent range to 66 percent based on discussions with OAQPS staff.[11] Electric arc furnace inhalable particulate fraction data was adjusted from 90 to 66 percent based on data in the Control Techniques document. In the other cases, no data were available to make any adjustments, so it was assumed that all of the particulate controlled at 15 micrometers or less were also all less than 10 micrometers. This may overestimate the amount of inhalable particulate controlled and, thus, improve inhalable particulate cost-effectiveness. However, given the absence of data to the contrary, this is the best estimate that can be made at this time.

After adjustments for inflation and the change in inhalable particle diameter, Table 9-6 gives the final estimates of the cost effectiveness on a total, inhalable, and fine particulate basis for the six stationary sources previously analyzed and the two sources addressed in Section B. They are listed in order of their inhalable particulate cost effectiveness, from best to worst. Also shown is some information on the control strategy on which the costs and emission reduction benefits are based for the previously analyzed sources.

IV. Discounted Cost Effectiveness for Mobile and Stationary Particulate Sources

A. Introduction

As discussed previously, it is desirable to compare emission sources based on their relative air quality impacts. Ideally, such an evaluation would account for the complex array of spatial and temporal characteristics associated with each emission source. Such an elaborate study is beyond the scope of this report, however. Instead, an attempt is made in this analysis to account for the relative ground level impacts of the various sources of particulate by evaluating dispersion characteristics. Emphasis is placed on ground level concentrations because the majority of the adverse effects from air pollution in populated areas violating the NAAQS for particulate matter occur due to ground level exposures (e.g., adverse effects on public health). Also, this is an important determinant of air pollution since compliance with the NAAQS is found by measuring ambient concentrations near ground level. Furthermore, this type of comparative evaluation has historically been used by EPA to model the relative contribution of various sources to ground level ambient concentrations of particular pollutants. (More will be said about these last two points later.)

Although such an evaluation of relative air quality provides a reasonable framework for comparing sources, it is nevertheless limited in its scope. For example, performing the analysis in this way ignores the location of these ground level concentrations associated with each source, particularly with respect to the number of people exposed and the local need for control (i.e., is the area in or out of compliance with the NAAQS). Unfortunately, this is a significant limitation since, for example, stationary sources can often be controlled on an individual basis (i.e., where the air quality problems are), while mobile sources cannot. This effect results in a relative inefficiency of the mobile source approach which cannot be factored in at this time. Thus, the conclusion of this cost-effectiveness comparison cannot be conclusive.

The comparison of cost effectiveness on an air qualtiy basis will be conducted in three steps. First, it will be necessary to determine an expression which relates the effect of various source characteristics on ground level particulate concentrations resulting from a given emission rate. Second, the pertinent source characteristics for the various sources under consideration here and the resultant air quality discount factors will also have to be determined. Third, once these factors have been determined, it will be possible to calculate discounted cost-effectiveness values for all sources which can then be compared with those for diesels.

A. Methodology for Evaluating the Ground Level Impact of Stationary Source Particulate Emissions

There are many characteristics unique to each source which can affect its relative contribution to ground level particulate concentrations. The meteorological conditions of the area, particle size and density, release height, and others can all affect dispersion. Given that 1) local meteorological conditions cannot be taken into account in a study of this breadth, and 2) this study is primarily concerned with particulate less than 10 microns in diameter (i.e., similar particle-related dispersion), the primary remaining factor affecting dispersion is release height.

In a recently released EPA document, an expression has been developed which provides a reasonable approximation of the dependence of the maximum ground level particulate concentration on effective release height.[12] This relationship is provided below:

$W = 10/H$ for H greater than 10 meters;

$W = 1$ for H less than or equal to 10 meters.

Where:

W = discount factor, maximum ground level particulate concentration relative to a ground level source,

H = effective release height, in meters (m).

This relationship is being used by OAQPS in their reconsideration of the NAAQS for particulate matter to relate the impact of various emission source controls on ambient particulate levels, which are measured near ground level, and compliance with the NAAQS. The general concept is also analogous to the use of source discount factors in rollback air quality modelling.

As would be expected, this equation showns an inverse relationship between maximum ground level contribution and effective release height; i.e., as release height increases, the maximum contribution from this source decreases.

B. Effective Release Heights and Discount Factors for Both Mobile and Stationary Sources

The effective release height for any emission source is equal to the sum of the physical stack height and the vertical height which the plume rises before significant horizontal dispersion occurs. While stack height is easily measured and fixed over time, plume rise varies according to source characteristics and meteorological conditions (e.g., stack gas temperature, exhaust gas flow rate, atmospheric stability, air temperature, wind velocity).

It is intuitively clear that the effective release height for diesel vehicles is less than 10 meters, and when evaluated in the equation above, yields the conclusion that diesel vehicles can be considered a ground level source (discount factor equal to 1.0). However, for stationary sources this may not be the case, and effective release height calculations are necessary.

A number of different models to calculate plume rise under various atmospheric stability conditions have been developed over the past 35 years. One approach which has gained widespread acceptance was developed by Briggs and will be used here to estimate the plume rise for the eight stationary sources under consideration.[13] As a further simplification, the Briggs formulae for a stable/near neutral atmosphere will be used in preference to those for an unstable atmosphere. It should be noted that this will tend to improve cost effectiveness (low cost-effectiveness values) of stationary source particulate controls, since particulate dispersion is significantly increased during increased atmospheric instability relative to that for neutral to stable atmospheres and the resulting ground-level impacts would be lowered.

The Briggs formulae (shown in Figure 9-1) require information on both source and atmospheric characteristics. Source characteristic values needed include the exhaust gas exit temperature and exhaust gas volumetric flow rate. These are shown in Table 9-7 along with their sources. Atmospheric conditions needed include the ambient air temperature, wind velocity, and atmospheric vertical temperature gradient at the stack exit. The choice to use an atmosphere with stable to near neutral characteristics will dictate values for these conditions. The values used here are -2°C/305 m for the ambient air temperature lapse rate, 288°K for the ambient air temperature at the stack exit, and 5 m/sec for the wind speed at the stack exit. These are fairly typical values for a midwestern U.S. city under stable to near neutral conditions, based on the ICAO U.S. standard atmosphere. The resultant plume rise heights are also shown in Table 9-7.

The effective release height is the sum of the stack height and the plume rise. Typical stack heights for the sources/control systems under consideration are given in Table 9-7. When these terms are added for the sources under consideration here, the effective release heights shown in Table 9-7 result. Using these effective release heights and the relationship given in the equation above, Table 9-7 gives the values of the weighting factor for the sources/control systems under construction here. Note that for diesel vehicles the weighting factor is 1.0 since the effective release height is less than 10 meters.

C. Air Quality Discounted Cost-Effectiveness Values

All that remains to be done to estimate cost effeciveness on an air quality basis is to divide the cost effectiveness values of Table 9-6 by the discount factors of Table 9-7. These discounted cost effectiveness values are shown in Table 9-8.

The figures in Table 9-8 show that after consideration of relative air quality effects, the base scenario is quite cost effective relative to stationary source controls regardless of the size of particulate examined. While the control of wet cement kilns is more cost effective than diesel particulate control across the board, only one other source is significantly more cost effective on a TSP basis (industrial boilers). No other sources are more cost effective on a fine

Figure 1

Plume Rise Calculation Equations

1. $$h = 2.3 \left(\frac{F}{Us}\right)^{1/3}$$

2. $$F = \frac{g\,Q\,(Ts - Ta)}{Ta}$$

3. $$S = \frac{g}{Ta}\,\frac{dT}{dz} + \frac{3C^\circ}{305m}$$

h = plume rise (meters).
F = bouyancy flux.
U = wind speed at stack exit (m/s).
s = stability parameter.
g = acceleration of gravity ($9.8\ m/s^2$).
Q = exhaust gas volumetric flow rate (m^3/s).
Ts = exhaust gas exit temperature (°K).
Ta = ambient air temperature at stack exit elevation (°K).
$\frac{dT}{dz}$ = ambient air temperature lapse rate.

Table 9-7

Source Characteristic Parameters, Plume Rise, Effective Release Height, and Weighting Factor

Source/Reference	Control System	Flow Rate $Q(Am^3/s)$	Stack Temp(°K)	Plume Rise(m)	Stack Height(m)	Effective Release Height(m)	Discount Factor (W)
Borax Furnace[5,9]	Scrubber	38	353	83	12	95	.105
Cement Kiln[5,9]	ESP	30	433	101	46	147	.068
Electric Utility[5]	ESP	533	400	242	175	417	.024
Industrial Boiler[5]	Baghouse	163	470	191	55	246	.041
Electric Arc Furnace[14]	Baghouse	62	346	95	19	114	.088
Rotary Lime Kiln[15]	ESP	3,000	474	509	30	539	.019
	Baghouse	800	405	281	25	306	.033
Kraft Furnace[16]	ESP	76	430	136	75	211	.047
Kraft Smelt Tank[16]	Scrubber	7,000	351	470	53	523	.019

Table 9-8

Summary
Air Quality Discounted Cost Effectiveness
Diesel Vehicles and Stationary Sources
($ per metric ton)*

Source	Particulate Size Basis Total	Inhalable	Fine
Wet Cement Kiln	1	810	2,000
HHDV**	2,850	2,850	2,850
LHDV**	5,420	5,420	5,420
MDV**	9,810	9,810	9,810
LDDT**	13,400	13,400	13,400
LDDV**	13,900	13,900	13,900
Kraft Smelt Tank	13,200	15,700	23,900
Electric Arc Furnace	10,500	16,400	16,500
Borax Fusinq Furance	14,600	19,250	21,700
Industrial Boiler	32,200	45,100	135,000
Kraft Recovery Furnace	35,700	45,600	65,000
Lime Kiln (Baghouse)	52,000	64,000	100,000
Electric Utility	52,250	75,200	170,500
Lime Kiln (ESP)	83,400	104,000	167,000

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* 1983 dollars.
Ranked according to inhalable particulate cost effectiveness.
Cost Effectiveness (Table 9-6) divided by Discount Factor (Table 9-7).

** Assumes presence of emissions averaging.

or inhalable particulate basis. (As mentioned earlier, the control of both fine and inhalable particulate are most important with respect to protecting the public health, the control of fine particulate is most important with respect to visibility, and the control of total particulate is most important with respect to soiling.) However, because of the limitations in the method used to determine the relative air quality impact of the various sources, these judgments cannot be made conclusively. At best, it can only be said that there is no evidence that diesel particulate control is not cost effective with respect to stationary source control.

To further place these figures in perspective, Table 9-9 shows estimates of annual emissions nationwide for most of the source categories listed in Table 9-8. However, the two tables do not match up exactly one-to-one. The emission estimates apply to entire industrial categories, while in a few cases (e.g., lime kilns and electric arc furnaces) the sources listed in Table 9-1 represent only a fraction of the industrial category emissions. Nonetheless, these emission estimates will be sufficient for our purposes here.

The nationwide emission estimates of Table 9-9 can be used to compare the potential for emission reduction from the stationary sources to that available for diesels. As can be seen, the base scenario will reduce nationwide emissions by roughly 120,000 metric tons per year in 1995. Only three of the stationary sources being considered here could potentially provide the same emission reduction: electric utilities, the cement industry, and industrial boilers. Given that the cement industry is predominantly located in rural areas,[17] only the remaining two sources can produce the same emission reduction where it is most needed. In addition, the impact of these sources on ground-level ambient concentrations relative to that of diesels must also be kept in mind.

VI. Summary

The cost effectiveness of the base scenario relative to the relaxed scenario has been estimated for five classes of diesels. For the purposes of comparing control between the diesel vehicle classes, cost-effectiveness values were determined on both a nationwide and urban basis, as well as for the control of total, inhalable and fine particulate. The cost effectiveness of controlling stationary source particulate emissions was also estimated. In order to compare these varied sources against the goal of improving air quality, emission control effectiveness was discounted according to the effective release height of the emission and its effect of dispersion. While this methodology accounts for source-specific dispersion

Table 9-9

Annual Nationwide Emission Rates by Source Category

Stationary Source (1981) [18]	Metric Tons Per Year
Electric Utilities	1,000,000
Cement Industry	460,000
Industrial Boilers	400,000
Concrete, Lime, Gypsum Industry	140,000
Pulp Mills	110,000
Iron and Steel Foundries	50,000
Borax Furnaces	Unavailable

On-Highway Diesels (best estimate sales)	Metric Tons Per Year
1980	140,000
1995 Relaxed Scenario	285,000
Base Scenario	166,000

effects, it does not account for important factors such as the location of the air quality improvement. This is a significant drawback, and prevents a fully appropriate comparison from being made.

The results of the analysis indicate that on an air quality basis the control of diesel particulate is cost effective relative to stationary source controls regardless of whether fine, inhalable, or total particulate are considered. However, due to the limitations of the methodology, the best that can be said at this time is only that there is no evidence that diesel particulate control is not cost effective with respect to available stationary source control and that the control of diesel particulate should not be avoided due to cost-effectiveness concerns. Between the subgroups of diesel vehicles, on an urban basis (the most appropriate) and assuming the presence of averaging, HHDVs are the most cost effective to control, followed by LHDVs, MDVs, LDDVs, and LDDTs.

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CHAPTER 10

SENSITIVITY

I. Introduction

This chapter contains a variety of analyses intended to address the sensitivity of the previous technical analyses to key assumptions that were made. The first analysis addresses the assumed levels of the LDV and LDT NOx standards. While the current NOx standard were presumed to continue indefinitely for ease of analysis, this is actually not likely to be the case. As additional NOx control tends to increase engine-out particulate levels, more stringent NOx standards would increase emissions under the relaxed scenario and increase the number of traps required under the base scenario. The cost effectiveness of trap application would also be affected.

The second analysis has a two-fold purpose. One, it addresses the assumption that the analysis of the base scenario, which only requires a minority of LDDVs and LDDTs to be equipped with traps, adequately addresses the economic viability (cost and cost effectiveness) of trap-oxidizer usage in general. Two, it expands the previous benefits analyses by estimating emissions (and, thus, other environmental effects) under the stringent particulate control scenario.

An intermediate control scenario for LDDVs and LDDTs is examined in the third analysis. This scenario, requiring the application of advanced non-trap technology, falls between the relaxed and the base scenarios in terms of stringency.

The fourth analysis addresses the possibility of using HDD emissions under the relaxed scenario as an estimate of uncontrolled emissions, which is usually desirable to present in a regulatory analysis.

The fifth analysis assesses the effect on diesel particulate emissions resulting from having no growth in future diesel sales. This provides a lower bound on the emission estimates which have been analyzed in the previous chapters.

The first two analyses will be presented together, as they overlap technically to a significant degree. The third, fourth, and fifth analyses will follow. It should be noted that these analyses will only address certain basic features of each scenario, such as fleet emissions, trap usage and cost effectiveness. More advanced aspects, such as exposure, cancer risk, and economic impact, are not presented. This was done because all of the benefits described in this study are proportional to fleet-wide emissions in a given calendar year, except for visibility effects, which are nearly proportional in the range being examined. Thus, the sensitivity of urban emissions in the sensitivity analyses indicates the same sensitivity in any other benefit category. A quantitative estimate of any or all benefits under one of the new scenarios being analyzed here can be determined simply by applying the ratio of fleetwide urban emissions to the estimate of benefits under one of the scenarios analyzed in the previous chapters. The same is true for economic impact, which is essentially proportional to the fraction of vehicles with traps.

II. Light-Duty NOx Standards and the Stringent Control Scenario

The previous chapters assumed that the NOx standard for LDVs and LDTs would remain at 1.5 and 2.3 g/mi, respectively, throughout the time period covered by this study. In this section, three additional sets of LDV/LDT NOx standards are investigated: 1) 1.0/1.2 g/mi, 2) 1.5/1.7 g/mi, and 3) 2.0/2.3 g/mi.

In addition, the previous chapters only addressed two control scenarios, the relaxed and the base scenarios. Here, a third scenario, the stringent scenario, will be examined. It consists of full, trap-based standards of 0.08 g/mi for LDDVs, 0.10 g/mi for LDDTs, and 0.10 g/BHP-hr for HDDs. The LDDV standard of 0.08 g/mi is that promulgated by California for the 1989 model year. The LDDT and HDD standards follow from this level in that they require the same percentage reduction from the base scenario.

Four key aspects of these scenarios will be addressed. The first aspect addressed will be manufacturers' corporate average particulate standard levels associated with the relaxed scenario under the three sets of NOx standards. The second and third aspects are directly related, the fraction of vehicles requiring traps under the base and stringent scenario, and urban particulate emissions in 1995 under the relaxed, base, and stringent scenarios under the various NOx standards. The fourth aspect will be the cost effectiveness of the base and stringent scenarios under the various NOx standards.

A. Manufacturers' Corporate Average Standard Level

The methodology used to estimate each manufacturer's current (relaxed scenario) corporate average standard level under NOx standards of 1.5 and 2.3 g/mi for LDVs and LDTs, respectively, was presented in Chapter 1. There, each engine configuration's low mileage particulate emission level was first adjusted for the NOx emission level under consideration.

This was accomplished through the use of estimated NOx/particulate tradeoff curves. The slope of the curve for small LDDV engines (1.6-1.8 liters displacement) was -0.033 for NOx values less than or equal to 1.35 g/mi and zero for NOx values greater than 1.35 g/mi. For medium LDDV engines (2.0 to 2.8 liters displacement), the slope of the curve was -0.20 for NOx values less than or equal to 1.35 g/mi and -0.10 for NOx values greater than 1.35 g/mi. For large LDDV engines, the slopes were -0.40 and -0.10 for NOx values less than and greater than 1.35 g/mi, respectively. The slopes of the NOx/particulate tradeoff curves were the same for LDDTs. However, small LDDTs have displacements from 1.6 to 2.3 liters and full-size LDDTs have displacements of 6.2 liters. There were no "medium" LDDTs.

Once each engine configuration's low-mileage particulate emission level was estimated, its particulate standard level was determined by multiplying the particulate emission level by its deterioration factor and the safety factor. Each manufacturer's engine configurations were then sales-weighted to give that manufacturer's corporate average standard level.

This methodology was repeated here for more stringent NOx standards (1.0/1.2 g/mi for LDVs and LDTs, respectively) and also for more relaxed NOx standards (2.0/2.3 g/mi LDVs/LDTs). Tables 10-1 and 10-2 show each manufacturer's corporate average particulate standard levels associate with the relaxed scenario for LDDVs and LDDTs under the various NOx standards. For LDDVs, going to a 1.0 g/mi NOx standard from a 1.5 g/mi NOx standard increases particulate emissions more than twice as much as going from a 2.0 g/mi NOx standard to a 1.5 g/mi NOx standard. The effect of moving to a 1.2 g/mi from a 1.7 g/mi NOx standard for LDDTs is small for small LDDTs but is dramatic for full-size LDDTs. The impact of moving from a 2.3 g/mi to a 1.7 g/mi NOx standard is negligible for small LDDTs but is measurable (18 percent increase) for full-size LDDTS. These impacts will reappear below when the effects of various NOx standards on urban emissions under the relaxed scenario are considered later in this section.

In applying this methodology to the stringent NOx standards (1.0 g/mi for LDVs and 1.2 g/mi for LDTs), the estimated NOx/particulate tradeoff curves were based on 1983 certification data, most of which were at higher NOx levels, and extrapolated to obtain these low NOx levels. There are currently California 1984 certification level data available for some but not all of the nationally certified engine families and an evaluation of the accuracy of the estimated values may be made on an engine family basis.[1] The estimation overestimated the particulate standard levels at low

Table 10-1

Relaxed Scenario Corporate Average Particulate Standard Levels for LDDVs (grams per mile)

Manufacturer	1.0 g/mi NOx Standard	1.5 g.mi NOx Standard	2.0 g/mi NOx Standard
General Motors	.50	.29	.25
Volkswagon	.21	.20	.20
Nissan	.29	.26	.25
Mercedes-Benz	.60	.42	.34
Isuzu	.22	.20	.20
Audi	.26	.20	.18
Peugeot	.36	.26	.21
Volvo	.41	.29	.24
Sales-Weighted Industry Wide Average	.42	.27	.24

Table 10-2

Relaxed Scenario Corporate Average Particulate Standard Levels for LDDTs (grams per mile)

Manufacturer	1.2 g/mi NOx Standard	1.7 g/mi Nox Standard	2.3 g/mi Nox Standard
Small LDDTS:			
Ford	.30	.29	.29
Isuzu	.33	.25	.25
Nissan	.37	.35	.35
Mitsubishi	.43	.39	.39
Toyota	.20	.19	.19
Volkswagon	.32	.31	.31
Toyo Kogyo	.30	.29	.29
Full-Size LDDTs:			
General Motors	.56	.34	.28
Sales-Weighted, Industry-Wide Average	.52	.33	.28

NOx standards for both LDDVs and LDDTs for the majority of the engine families that were certified under California's 1984 1.0 g/mi/1.2 g/mi NOx standards, by an average of approximately 15 percent. Thus, the corporate average particulate standard levels for the low NOx standards should be considered as upper limits, as can the resulting number of vehicles projected to require traps under this scenario, as discussed in the following section.

B. Percent of Trap-Equipped Vehicles

The methodology for calculating the percentage of each model year's LDDVs and LDDTs to be equipped with trap-oxidizing systems was also presented in Chapter 1. Basically, the number of vehicle grams per mile (veh-g/mi) of diesel particulate allocated to each manufacturer under the base scenario (i.e., particulate "averaging" standards of 0.20 and 0.26 g/mi for LDDVs and LDDTs respectively) was determined from manufacturer's projected 1985 sales. Then, the number of veh-g/mi of diesel particulate that would actually be emitted by each engine configuration under NOx standards of 1.5 and 2.3 g/mi for LDVs and LDTs, respectively, without traps were calculated. Finally, traps were applied to reduce each manufacturer's diesel particulate veh-g/mi to the allowable level which gave the percentage of each manufacturer's production that would need to be equipped with traps.

Tables 10-3 and 10-4 show the percentage of each manufacturer's LDDV and LDDT production (and that of the overall fleet) that would need to be equipped with traps under the base and stringent scenarios for the three sets of NOx standards. Under a 1.0 g/mi LDV NOx standard, the percentage of the LDDV fleet which would require traps under the base scenario would more than double from that required under a 1.5 g/mi NOx standard. Conversely, a 2.0 g/mi NOx standard would reduce the requirement for traps by almost half. The stringent scenario would require the LDDV fleet to be trap-equipped as follows: 1) nearly all under a 1.0 g/mi NOx standard (95 percent); 2) 82 percent under a 1.5 g/mi NOx standard; and 3) 72 percent under a 2.0 g/mi NOx standard. The trap fractions for LDDTs follow very closely those for LDDVs.

As explained in Chapter 1, 100 percent of HDDs are equipped with traps under the base scenario without averaging. With averaging the percentage of traps would drop to about 70 percent. Under the stringent scenario, essentially all HDDs are equipped with traps, with or without averaging.

Table 10-3

Percentage of LDDVs Requiring Traps Under Various NOx and Particulate Standards, (g/mi)

	Stringent Part., 0.08			Base Part., 0.20		
Manufacturer	1.0 NOx	1.5 NOx	2.0 NOx	1.0 NOx	1.5 NOx	2.0 NOx
General Motors	100	81	73	61	27	15
Volkswagen	83	78	67	6	0	0
Nissan	89	88	75	33	26	23
Mercedes-Benz	100	96	79	80	55	45
Isuzu	82	77	67	9	0	0
Audi	84	77	62	28	2	0
Peugeot	96	87	64	55	30	5
Volvo	100	93	71	58	34	16
Sales-Weighted Industry-Wide Percentage	95	82	72	48	22	14

Table 10-4

Percentage of LDDTs Requiring Traps Under Various NOx and Particulate Standards, (g/mi)

	Stringent Part., 0.10			Base Part., 0.26		
Manufacturer	1.2 NOx	1.7 NOx	2.3 NOx	1.2 NOx	1.7 NOx	2.3 NOx
General Motors	98	85	77	63	26	7
Volkswagen	79	78	78	14	15	15
Nissan	89	87	87	37	32	32
Isuzu	85	74	74	26	0	0
Ford	82	80	80	16	12	12
Mitsubishi	92	89	89	47	40	40
Toyota	55	55	55	0	0	0
Toyo Kogyo	81	80	80	15	11	11
Sales-Weighted Industry-Wide Percentage	95	83	77	56	24	8

C. 1995 Urban Diesel Particulate Emissions Under Various NOx Standards

Having calculated industry-wide particulate standard levels and percentages of traps required under each scenario, the 1995 particulate emission factors for LDDVs and LDDTs can be calculated. As explained in Chapter 2, the 1995 particulate emission factors for LDDVs or LDDTs of a specific model year are calculated using the age distribution of the in-use fleet, the percentage of that model year's fleet equipped with traps, the average non-trap emission level of those vehicles which are equipped with traps, the particulate standard, and the annual trap-failure rate (i.e., 1.5 percent per year).

The particulate emission factors for 1961-86 model year LDDVs and LDDTs remain the same as those in Chapter 2, due to the fact that all regulatory changes are assumed to occur in 1987. For the relaxed scenario, the emission factors for model year 1987-95 are the sales-weighted industry-wide averages shown in Tables 10-1 and 10-2. For the base and stringent scenarios, the 1987-95 emission factors are essentialy the same for all NOx standards since the presence of a standard requiring control sets the emission level regardless of the starting point. However, these particulate emission factors are slightly different for each NOx standard, because both the fleet-wide trap fraction and the average non-trap particulate emission levels of those vehicles with traps change as the applicable NOx standard changes. When a trap-oxidizer system fails, the particulate emission level that the vehicle reverts to is different under each NOx standard because of the previously described NOx/particulate tradeoff. For vehicles with properly operating traps, the emission factors are the same.

These 1995 particulate emission factors were combined with the vehicle miles traveled (VMT) breakdown by model year and the diesel sales fractions (for both best estimate and worst case sales) to yield weighted fleet-wide particulate emission factors for each set of NOx standards. Again, this methodology is fully described in Chapter 2. To obtain 1995 urban diesel particulate emissions, the weighted particulate emission factors were multiplied by the total 1995 VMT for each vehicle class and by the urban fraction of VMT for each vehicle class (i.e., 0.594 and 0.488 for LDVs and LDTs, respectively).

Table 10-5 presents the 1995 urban diesel particulate emissions for best estimate and worst case diesel sales under the relaxed and base control scenarios and combinations of the various NOx standards. Table 10-6 shows the relative contribution of each vehicle type to the totals of Table 10-5.

Table 10-5

1995 Urban Diesel Particulate Emissions Under Various NOx Standards (metric tons)

	LDV NOx = 1.0 g/mi LDT NOx = 1.2 g/mi		LDV NOx = 1.5 g/mi LDT NOx = 1.7 g/mi		LDV NOx = 1.5 g/mi LDT NOx = 2.3 g/mi		LDV NOx = 2.0 g/mi LDT NOx = 2.3 g/mi	
Vehicle Type	Relaxed Scenario	Base Scenario	Relaxed Scenario	Base Scenario	Relaxed Scenario	Base Scenario	Relaxed Scenario	Base Scenario
				Best Estimate Diesel Sales				
LDDV	36,800	19,700	24,600	19,100	24,600	19,100	22,200	19,000
LDDT	21,800	11,700	14,100	11,500	12,200	11,500	12,200	11,500
Total*	135,600	72,200	115,700	71,400	113,800	71,400	111,400	71,300
				Worst Case Diesel Sales				
LDDV	84,000	43,800	55,300	42,400	55,300	42,400	49,600	42,200
LDDT	33,900	18,000	21,800	17,600	18,800	17,600	18,800	17,600
Total*	205,900	108,000	165,100	106,200	162,000	106,100	156,400	105,900

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* Totals include MDV/LHDV and HHDDV emissions of 9,400 and 67,600 (relaxed), and 4,700 and 36,100 (Base) for best estimate sales and 18,700 and 69,200 (relaxed) and 9,100 and 37,000 (Base) for worst case sales. These are not shown since they are the same regardless of LDV and LDT NOx standards.

Table 10-6

Relative Contribution of 1995 Urban Diesel Particulate Emissions Under Various NOx Standards (percent)

Vehicle Type	LDV NOx = 1.0 g/mi LDT NOx = 1.2 g/mi Relaxed Scenario	Base Scenario	LDV NOx = 1.5 g/mi LDT NOx = 1.7 g/mi Relaxed Scenario	Base Scenario	LDV NOx = 1.5 g/mi LDT NOx = 2.3 g/mi Relaxed Scenario	Base Scenario	LDV NOx = 2.0 g/mi LDT NOx = 2.3 g/mi Relaxed Scenario	Base Scenario
Best Estimate Diesel Sales								
LDDV	27	27	21	27	21	27	20	26
LDDT	16	16	12	16	11	16	11	16
MDV/LHDV	7	7	8	6	8	6	8	7
HHDV	50	50	59	51	60	51	61	51
Total*	100	100	100	100	100	100	100	100
Worst Case Diesel Sales								
LDDV	40	40	33	40	34	40	31	40
LDDT	16	17	13	16	12	16	12	16
MDV/LHDV	10	9	12	9	11	9	12	9
HMDV	34	34	42	35	43	35	45	35
Total*	100	100	100	100	100	100	100	100

The results shown in Table 10-5 indicate that 1995 urban diesel particulate emissions under the relaxed scenario do not change substantially from those evaluated in Chapter 2 (i.e., 1.5/2.3 g/mi NOx standards) except for the most stringent 1.0/1.2 g/mi NOx standards. For the 1.0 g/mi NOx standard, LDDV emissions increase by 50 percent as compared to a 1.5 g/mi NOx standard. For the 1.2 g/mi NOx standard, LDDT emissions increase by 79 percent as compared to a 2.3 g/mi NOx standard. Total 1995 urban diesel particulate emissions increase by 19 percent under the 1.0/1.2 g/mi set of NOx standards as compared to the 1.5/2.3 g/mi set.

Under the base scenario, the changes with NOx standards are less significant. The 1995 LDDV urban diesel particulate emissions increase only 3 percent under a 1.0 g/mi NOx standard as compared to a 1.5 g/mi NOx standard. For the other changes in the LDDV and LDDT NOx standards, the situation is similar, with very little change in emissions occurring.

The results of Table 10-6 are similar to Table 10-5 in that the only NOx standards causing strong difference from the main analysis are the 1.0/1.2 g/mi NOx standards under the relaxed scenario. The contribution of LDDVs and LDDTs under best sales estimate to total 1995 urban diesel particulate emissions increases from 21 to 27 percent, and from 11 to 16 percent, respectively, under the more stringent set of NOx standards. The other vehicle types (i.e., MDV/LHDV and HHDV) decrease their relative contribution with HHDV's share decreasing the most (from 60 to 50 percent). The results are similar for the worst case diesel sales situation.

Under the stringent scenario, there is little difference among NOx standards (see Table 10-7). Overall, the breakdown under the stringent scenario is between that under the relaxed and base scenarios.

The decrease in urban emissions obtained under the stringent scenario versus the relaxed scenario is between 56 and 72 percent for all vehicle types with the total decrease being 65 percent. Compared to the base scenario, the stringent scenario reduces total emissions by 43-46 percent, with the change in each vehicle class being 37-54 percent.

Table 10-8 compares urban emissions under the three control scenarios coupled with NOx standards of 1.0/1.2 g/mi to previous diesel particulate studies. As can be seen, projections for the base scenario are virtually the same as that projected in 1979-80 for the same standards (controlled scenario). However, even under the stringent NOx standards, emissions under the current relaxed scenario are well below the uncontrolled levels projected in previous analyses.

Table 10-7

1995 Urban Diesel Particulate Emissions Under the Stringent Scenario (metric tons)

Vehicle Type	1.0/1.2 g/mi NOx	1.5/1.7 g/mi NOx	2.0/2.3 g/mi NOx	Relative Contribution (%)	Reduction From Base (%)
Best Estimate Diesel Sales					
LDDVs	10,700	10,000	9,500	24	48
LDDTs	5,900	5,600	5,400	13	53
MDDV/LHDDVs	2,700	2,700	2,700	7	44
HHDDVs	22,600	22,600	22,600	56	38
Total	41,900	40,900	40,200	100	43
Worst Case Diesel Sales					
LDDVs	22,500	20,900	19,700	36	51
LDDTs	8,900	8,500	8,100	14	54
MDDV/LHDDVs	5,000	5,000	5,000	9	46
HHDDVs	23,100	23,100	23,100	41	37
Total	59,500	57,500	55,900	100	46

Table 10-8

Comparison of Current Urban Emission Estimates Under Various NOx Standards* to Urban Emission Estimates of Previous Studies

Scenario	1995 Urban Emissions Under LDV and LDT NOx Standards of 1.0 and 1.2 g/mi (metric tons)
Best Estimate Diesel Sales	
1979-80 Uncontrolled	239,000
Relaxed	137,000
Base	72,900
1979-80 Controlled	71,000
Stringent	42,300
Worst Case Diesel Sales	
1979-80 Uncontrolled	287,000
Relaxed	207,300
Base	108,600
1979-80 Controlled	85,000
Stringent	59,900

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* The NOx standard scenarios of LDV = 1.5/LDT = 1.7, and LDV = 2.0/LDT = 2.3 g/mi are not shown because all "relative reductions" are less than 4 percentage points different than the 1.5/2.3 g/mi case.

D. Cost Effectiveness

The cost effectiveness for LDDs and HDDs under the base scenario was already determined in Chapter 9. Tables 9-1, 9-2, and 9-3 of that chapter show the development of those cost-effectiveness values.

In this study, cost effectiveness is the annualized cost per vehicle divided by the annual emission reduction per vehicle, both relative to the relaxed scenario and on a fleet-average basis. The fleet-average annualized cost is a straight-forward annualization of the fleet-average lifetime costs using a 10 percent discount rate. The fleet-average lifetime costs are a function of the lifetime costs of trap-equipped vehicles of various sizes, the trap-equipped fraction of each vehicle size category, and the relative sales of each vehicle size category. The lifetime trap-oxidizer system costs for different size vehicles and the relative sales of these vehicle sizes were described in Chapter 8. The trap-equipped fractions of the LDDV and LDDT fleets under various NOx standards were estimated in Section IIB of this chapter.*

The determination of annual emission reductions was explained in Chapter 2. Basically, the annual emission reduction per vehicle is approximately the reduction in the vehicle's emission rate at half-life (compared to the relaxed scenario) multiplied by the lifetime-average annual VMT. The effect of trap failures is included in the vehicular emission rate.

Table 10-9 compares the cost effectiveness of the various LDD particulate control scenarios under different NOx standards. Table 10-10 compares the cost effectiveness of the various HDD particulate control scenarios. These tables include the fleet-average annualized cost per vehicle, the annual emission reductions per vehicle, and the urban cost effectiveness (as described in Chapter 9). Table 10-9 also shows the trap-equipped fraction for LDDs (assumed to be 100 percent for HDDs, except where averaging is applicable the fraction is reduced to 70 percent).

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* It is assumed in this analysis that for LDDVs, large vehicles are first equipped with traps, followed by medium vehicles, and then small vehicles until the trap-equipped fraction is met. Similarly, for LDDTs, full-sized LDDTs are first equipped with traps, and then small LDDTs, until the trap-equipped fraction is met.

Table 10-9

LDDV and LDDT Cost-Effectiveness Values Under Various Particulate Control Scenarios and NOx Standards ($/metric ton)

	Base Scenario			Stringent Scenario		
	NOx=1.0/ 1.2 g/mi	NOx=1.5/ 1.7 g/mi	NOx=2.0/ 2.3 g/mi	NOx=1.0/ 1.2 g/mi	NOx=1.5/ 1.7 g/mi	NOx=2.0/ 2.3 g/mi
Percent Vehicles Equipped with Traps						
LDDVs	48%	22%	14%	95%	82%	72%
LDDTs	56%	24%	8%	95%	83%	77%
Fleet Average Annualized Cost Per Vehicle*						
LDDVs	$18.85	$9.19	$5.73	$35.27	$30.79	$27.85
LDDTs	$20.77	$8.83	$2.81	$34.56	$31.24	$28.34
Annual Emission Reduction Per Vehicle (metric tons)**						
LDDVs	2.07×10^{-3}	6.60×10^{-4}	3.90×10^{-4}	3.15×10^{-3}	1.77×10^{-3}	1.54×10^{-3}
LDDTs	2.65×10^{-3}	7.00×10^{-4}	2.10×10^{-4}	4.10×10^{-3}	2.20×10^{-3}	1.77×10^{-3}
Cost Effectiveness ($/metric ton)						
LDDVs	$9,100	$13,900	$14,700	$11,200	$17,400	$18,100
LDDTs	$7,800	$12,600	$13,400	$8,400	$14,200	$16,000
Urban Cost Effectiveness ($/metric ton)						
LDDVs	$15,400	$23,400	$24,700	$18,900	$29,300	$30,800
LDDTs	$16,100	$25,800	$27,400	$17,300	$29,100	$32,800

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* Based on estimated sales fractions of 29, 37, and 32 percent for large, medium, and small LDDVs, respectively; trap-oxidizer systems fitted to these vehicles have an average lifetime cost of $219, $234, and $266, respectively. Small and full-sized LDDTs are estimated at 34 and 66 percent of sales respectively, with trap-oxidizers system lifetime costs of $229 and $252, respectively.

** Based on estimated annualized travel of 10,000 miles and 10,900 miles for LDDVs and LDDTs, respectively; reductions are compared to relaxed scenario.

Table 10-10

HDD Cost-Effectiveness Values
Under Various Control Scenarios ($/metric ton)

	Base Scenario	Stringent Scenario
Fleet-Average Annualized Cost Per Vehicle*		
MDVs	$ 84	$ 84
LHDVs	$132	$132
HHDVs	$228	$228
Annual Emission Reduction Per Vehicle (metric tons)**		
MDVs	0.0060	0.0086
LHDVs	0.0171	0.0246
HHDVs	0.0560	0.0799
Cost Effectiveness ($/metric ton)		
MDVs	$14,000	$9,800
LHDVs	$ 7,700	$5,400
HHDVs	$ 4,100	$2,800
Cost Effectiveness, With Averaging ($/metric ton)***		
MDVs	$9,800	--
LHDVs	$5,400	--
HHDVs	$2,800	--
Urban Cost Effectiveness, With Averaging ($/metric ton)		
MDVs	$20,000	$20,000
LHDVs	$11,000	$11,000
HDVs	$11,000	$11,000

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* Assumes all HDDVs are equipped with traps, unless averaging is used. Trap-oxidizer systems for MDVs, LHDVs, and HHDVs have an average lifetime cost of $472, $425, and $1,516, respectively.

** Based on estimated annualized travel of 13,750, 23,300, and 50,100 miles for MDVs, LHDVs, and HHDVs, respectively; reductions are compared to relaxed scenario.

*** Averaging affects the base scenario only; average fleetwide costs are estimated to decrease by 30 percent (i.e., 70 percent of HDDs equipped with traps).

Table 10-9 shows that under a given set of standards, cost-effectiveness of LDDT control ranges between $1000-3000 per metric ton less than that for LDDVs, meaning that LDDT control is slightly more cost effective. More importantly, the table also shows that a given particulate scenario becomes less cost effective with higher NOx standards. For example, control is noticeably less cost effective when the NOx standards change from 1.0/1.2 g/mi to 1.5/1.7 g/mi. This is due to the fact that stringent NOx controls raise engine-out particulate levels and increase the degree of control provided by adding a trap.

Under a given NOx standard, the stringent scenario is moderately less cost effective than the base scenario. The difference, which ranges between 7 and 27 percent, is to be expected, since the additional traps being applied under the stringent scenario are being applied to vehicles with lower engine-out particulate levels, thus providing less control. Trap costs, on the other hand, are relatively constant.

For heavy-duty diesels (Table 10-10), cost effectiveness improves from the lighter to the heavier vehicles. While the emission reductions for the various HDD classes are the same on a percentage basis, they are greater for the heavier vehicles on an absolute basis (due to greater absolute emission rates and greater annual VMT). These effects more than compensate for the increase in trap cost with vehicle size and the lower urban VMT fraction of Class VII-VIII HDDs.

Without averaging, the base scenario for HDDs is less cost effective than the stringent scenario. Without averaging, the base scenario, like the stringent scenario, requires all HDDs to be equipped with traps. It was assumed that traps under the base scenario would only be as efficient as needed, but would cost the same as traps under the stringent scenario. Thus, the costs of both scenarios are the same, but the emission reduction under the base scenario is less. Thus, the higher cost-effectiveness value of the base scenario.

With averaging, the cost effectiveness of the base and stringent scenarios becomes the same. This is to be expected. Trap costs and efficiency under the two scenarios are assumed to be the same. The only difference between the two scenarios is that only 70 percent of all HDDs are equipped with traps under the base scenario, while all HDDs are trap-equipped under the stringent scenario. However, this difference affects both costs and emission reductions. Thus, cost effectiveness remains constant.

In general, particulate control for HDDs is more cost effective than that for LDDs when compared under the same scenario.

III. The Intermediate Control Option

Between the relaxed scenario (no control aside from that already applied) and the base scenario (some traps required as a control method) there is a third scenario: the intermediate control option. In this option a modest degree of control may be obtained via predominantly non-trap technology. For example, electronic fuel injection and sophisticated electronic exhaust gas recirculation (EGR) systems (to reduce the negative impact of stringent NOx standards) appear to be available and able to provide some control. There is also the possibility that certain high-emitting engine lines may be dropped in this timeframe due to fuel economy and other pressures. Finally, there is some indication that oxidation catalyst technology coupled with sohisticated throttle control can be applied to some diesels to substantially reduce particle-bound organics. The effect these non-trap control techniques could have in reducing the current corporate average particulate standard levels, at both the 1.0/1.2 g/mi and 1.5/1.7 g/mi NOx standard levels, will be discussed in this section.

It appears almost certain that GM will drop their 5.7-liter engine by 1987 of their own accord. (GM has already dropped this engine from its 1984 California model line and has indicated the engine will be eliminated from the Federal market in 1986.) Assuming that the presently equipped 5.7-liter engine vehicle will receive GM's 4.3-liter engine, GM's average particulate standard would drop from 0.29 g/mi to 0.16 g/mi at the 1.5 g/mi NOx standard and from 0.50 g/mi to 0.34 g/mi at the 1.0 g/mi NOx standard. These large reductions in GM's corporate average emission levels under both NOx levels would result in substantial reductions to the total LDDV particulate emissions, due to GM's large share (60 percent) of the total LDDV estimated sales. The sales-weighted industry-wide average particulate standard levels, as listed in Table 10-1, would change from 0.27 g/mi to 0.20 g/mi at the 1.5 g/mi NOx standard and from 0.42 g/mi to 0.32 g/mi at the more stringent 1.0 g/mi NOx standard, a 25 and 24 percent reduction, respectively.

Engine-out control techniques include EGR systems and electronic fuel injection. Sophisticated EGR systems are available and already being applied on a few 1984 vehicles in California to comply with its 1.0 g/mi NOx standard. These systems should be able to reduce the particulate penalty associated with the stringent NOx standard by one-half. While the effect of electronic injection is uncertain, it should provide a benefit for the highest emitting engines; at least improving the NOx/particulate trade-off.

Some data on catalyst technology coupled with intake-air throttling shows that it can remove most of the organics associated with the particulate, which represent 10-20 percent of total particulate mass. Since particulate is not permanently stored by these catalysts, no regeneration is needed and the practical problems associated with trap-oxidizers are avoided. However, one remaining question with this technology is sulfate production. Also, it is not certain whether catalysts would be effective on all LDD models. At least one manufacturer (Volkswagen), which only requires a small emission reduction to meet the current 1985 standards and should be interested in catalysts, appears to be concentrating all of its efforts on traps. Thus, this analysis will not presume the availability of catalysts under this option.

The identification of standards actually achievable with these techniques requires a manufacturer by manufacturer analysis, due to their different starting points and the fact that some of these reductions are only applicable to certain manufacturers. Also, the two NOx standards (1.0 and 1.5 g/mi for LDDVs and 1.2 and 1.7 g/mi for LDDTs) must be discussed separately due to the differences in the control techniques' effects on particulate emissions at these different NOx levels.

Beginning with LDDVs at the more stringent NOx standard and GM, the application of sophisticated EGR systems and the discontinuation of their 5.7-liter engine line would reduce their average emission level to roughly 0.23 g/mi. Use of electronic fuel injection could reduce this level further.

Mercedes-Benz, on the other hand, appears to be in a very different position. Since it has not projected the discontinuation of any of its engine lines, nor has there been any indication that simple changes in basic engine design are forthcoming, emissions of its 3.0-liter engine will not be in line with those of the other manufacturers, though its new 2.2-liter engine has relatively low emissions. Mercedes-Benz has actually indicated to California that, in 1985, they plan to equip their 3.0-liter engines with traps to meet the 0.4 g/mi California particulate standard coupled with the 1.0 g/mi NOx standard. Improved EGR systems should be able to reduce their corporate average from 0.60 g/mi to at best 0.40 g/mi. Electronic injection could further reduce this figure, but probably not below 0.35 g/mi. At the same time, Mercedes-Benz' trap development program appears to be more advanced than those of the other manufacturers and they appear ready to implement traps in 1985. Also, one cannot rule out the use of major engine modifications to bring its 3.0-liter engine emissions in line with those of other manufacturers. Overall, it appears most likely that Mercedes-Benz would probably require traps at or below 0.40 g/mi, but will also be in good position to do this.

Concerning the other LDDV manufacturers, improved EGR systems alone should reduce their average emission level to 0.30 g/mi or below, except possibly for Peugeot, due to a single high-emitting vehicle configuration. Electronic fuel injection and other engine modifications may be able to reduce these levels to 0.25 g/mi.

Overall, these techniques could reduce particulate emissions roughly 30-40 percent from the relaxed scenario's 0.42 g/mi sales-weighted industry-wide average value at a 1.0 g/mi NOx standard. Thus, an LDDV standard range of 0.25 to 0.30 g/mi should be achievable without many traps, except for Mercedes-Benz. In the case of Mercedes-Benz, it does not appear reasonable to limit the achievable emission standard to 0.40 g/mi or even higher based on a single manufacturer or vehicles no larger or smaller than other manufacturers. They and possibly others would simply have to apply traps to a limited number of vehicles in order to obtain the final degree of control.

At the higher NOx standard level of 1.5 g/mi for LDDVs, the engine improvements just discussed will not have as large an effect on particulate emissions as just seen at the more stringent NOx standard. The particulate/NOx tradeoff, as discussed in the technology chapter, levels out as the NOx level increases from the 1.5 g/mi NOx standard level. The particulate emission reduction due to engine improvements combined with the discontinuation of the GM 5.7-liter engine would result in an overall LDDV standard of approximately 0.18-0.20 g/mi. At this level some traps will be necessary for the higher-emitting engines of Mercedes Benz and Peugeot although not to the extent needed at the 1.0 g/mi NOx standard.

The application of these non-trap control techniques to LDDTs also reduces their average particulate emission levels from those shown in Table 10-2, although to a far less extent than the LDDV emission reduction. Unlike LDDVs, there are no discontinuations projected for any of the higher-emitting engine lines. These techniques could reduce the sales-weighted industry-wide average values from 0.52 g/mi to approximately 0.35 g/mi and from 0.33 g/mi to approximately 0.30 g/mi, for NOx standards of 1.2 and 1.7 g/mi, respectively.

The sales-weighted industry-wide particulate levels under the intermediate control option are shown in Table 10-11. This table includes both LDDVs and LDDTs corporate average particulate standard levels at the two NOx standards. For the LDDVs, the higher value of the standard range was included, as a conservative projection. The percentage of vehicles requiring traps is also shown in Table 10-11. Applying the

Table 10-11

The Effects of Advanced Control Technology Improvements on the Sales-Weighted Industry-Wide Average Particulate Standard Levels

	LDDV		LDDT	
Scenario	1.0 NOx Std	1.5 NOx Std	1.2 NOx Std	1.7 NOx Std
Intermediate				
Standard Level, g/mi.	0.30	0.18-0.20	0.35	0.30
Percentage of Vehicles Requiring Traps	1-2	1-2	9-16	0
Base				
Standard Level, g/mi	0.20	0.20	0.26	0.26
Percentage of Vehicles Requiring Traps	10-20	1-2	41	16

advanced control technologies described above and also including the effects of dropping the GM 5.7-L engine will require approximately one to two percent of LDDV engines to be equipped with traps under the intermediate scenario for both a 1.0 and 1.5 g/mi NOx standard. (The more stringent NOx standard will require approximately an equal amount of traps as the less stringent 1.5 NOx standard due to its higher corporate average particulate standard level under the intermediate scenario.) Applying the advanced control technologies to LDDT results in 9-16 percent of the LDDTs at the 1.2 g/mi NOx standard requiring traps; at the 1.7 g/mi NOx standard, no traps are necessary.

In addition to using only traps as a control method under the base scenario, it is of interest to project the percentage of vehicles requiring traps if the advanced control technologies of the intermediate scenario are also applied under this 0.20/0.26 g/mi LDDV/LDDT particulate standard scenario. Ten to 20 percent of the LDDVs will be equipped with traps under the 1.0 g/mi NOx standard, while one to two percent will require traps under the 1.5 g/mi NOx standard. It has also been assumed that the 5.7-L engine will be dropped from the GM engine line. With the advanced control technology improvements applied to LDDTs, 45 percent of the full-size LDDTs and 31 percent of the small LDDTs will require traps at the 1.2 g/mi NOx standard. (Forty-one percent of all LDDTs will require traps.) At the 1.7 g/mi NOx standard, 18 percent of the small LDDTs and 11 percent of the small LDDTs will be trap-equipped. (Sixteen percent of all LDDTs will be trap-equipped.) These percentages of vehicles requiring traps under the base scenario are shown in Table 10-11.

IV. Comparison of Uncontrolled and Controlled HDD Emissions

Urban diesel particulate emissions from HDDs were estimated for the relaxed control scenario in Chapter 2. However, the corresponding values for a completely uncontrolled HDD fleet were never derived in that chapter because such a strategy is not considered to be a viable option. Nevertheless, it is of interest to know what future urban HDDV emissions would be if the fleet were totally uncontrolled so that the benefits of the relaxed control scenario in particular can be placed in perspective.

As indicated in Chapter 1, uncontrolled HDDVs are estimated to emit particulate at a rate of 0.7 g/BHP-hr throughout their lifetime. Using the methodology of Chapter 3, the resulting vehicular emission factors are shown in Table 10-12. Table 10-13 presents the 1995 urban particulate emissions for the various HDDV scenarios using best estimate

Table 10-12

Uncontrolled HDDV Emission Factors By Model Year (g/mi)

	MDV		LHDV	HHDV
Model Year	Class IIB	Classes III-V	Class VI	Classes VII-VIII
1995	0.930	1.212	1.611	2.489
1994	0.930	1.216	1.611	2.492
1993	0.930	1.221	1.611	2.497
1992	0.930	1.221	1.611	2.501
1991	0.930	1.221	1.611	2.506
1990	0.930	1.234	1.611	2.509
1989	0.936	1.235	1.612	2.514
1988	0.943	1.235	1.612	2.519
1987	0.951	1.239	1.614	2.530
1986	0.967	1.239	1.614	2.531
1985	0.968	1.243	1.614	2.550
1984	0.977	1.247	1.617	2.558
1983	0.993	1.247	1.620	2.561
1982	1.002	1.247	1.628	2.577
1981	1.003	1.252	1.651	2.587
1961-80	1.014	1.256	1.651	2.597

Table 10-13

HDDV Urban Emissions in 1995
Best Sales Estimates (metric tons per year)

Uncontrolled	Relaxed	Base	Stringent
88,000	77,000	40,800	25,300

sales projections. Relative to the uncontrolled scenario, the relaxed scenario would reduce particulate emissions about 12 percent, the base scenario about 54 percent, and the stringent scenario about 71 percent.

V. Effect of Changes in Diesel Sales Projections on Urban Particulate Levels

Throughout this report, diesel particulate emissions were evaluated using EPA's best and worst diesel sales projections. The "best case" projection is based on a reversal of very recent conditions (where LDDV sales have been decreasing) and shows moderate growth in sales. The "worst case" projection represents a significant (or maximum) growth in diesel sales that could result from another oil crisis. Therefore, the worst case is an upper bound on future diesel sales. Of course, it is also possible that the demand for diesel-fueled vehicles will not continue to increase if petroleum supplies remain abundant and the fuel price does not escalate. In order to indicate the effect on diesel particulate of this latter sales projection, a "no growth case" will be evaluated.

In constructing the no growth diesel projection, LDDV and LDDT historical diesel and total sales fractions were used for model years 1969-83. Future model year diesel sales fractions were then found by simply continuing the 1983 model year values through the 1995 model year. For HDDVs, historical data were similarly used for model years 1969-82. The 1983 model year sales fraction was determined from projections made by Energy and Environmental Analysis, Inc.[2] and were extrapolated through the 1995 model year. The resulting no growth sales projection is shown in Table 10-14.

Diesel particulate emissions under the no growth case for any calendar year can now be calculated by combining the respective diesel sales fractions with the other inputs as described in Chapter 2. To illustrate the effect of using this alternative growth projection, Table 10-15 contains a comparison of urban diesel particulate emissions for the base scenario under this sales estimate to those associated with the best estimate projection. As shown, in the 1995 calendar year, the overall effects of no growth in diesel sales is a 36 percent reduction in projected emission levels associated with the best estimate. Individually, the largest change is shown for LDDTs (i.e., a 71 percent reduction), while the smallest change is shown for HHDVs (i.e., a 13 percent reduction). The percent change within each vehicle category shown here for the base scenario also will be similar for the other control scenarios if the no growth sales projection was used in place of the best sales estimate.

Table 10-14

Diesel Fraction of Total Sales for LDDs and HDDVs--No Growth Estimate

Model Year	LDDV	LDDT	Class IIB	Classes III-V	Class VI	Classes VII-VIII
1995	.030	.060	.162	.162	.377	.810
1994	.030	.060	.162	.162	.377	.810
1993	.030	.060	.162	.162	.377	.810
1992	.030	.060	.162	.162	.377	.810
1991	.030	.060	.162	.162	.377	.810
1990	.030	.060	.162	.162	.377	.810
1989	.030	.060	.162	.162	.377	.810
1988	.030	.060	.162	.162	.377	.810
1987	.030	.060	.162	.162	.377	.810
1986	.030	.060	.162	.162	.377	.810
1985	.030	.060	.162	.162	.377	.810
1984	.030	.060	.162	.162	.377	.810
1983	.030	.060	.162	.162	.377	.180
1982	.040	.070	.074	.108	.214	.928
1981	.061	.060	.037	.054	.164	.918
1980	.034	.034	.000	.000	.114	.910
1979	.028	.028	.000	.000	.114	.890
1978	.009	.009	.000	.000	.078	.880
1977	.004	.005	.000	.000	.070	.850
1976	.003	.003	.000	.000	.042	.830
1975	.003	.002	.000	.004	.032	.730
1974	.003	.000	.000	.001	.016	.770
1973	.003	.000	.000	.003	.016	.780
1972	.003	.000	.000	.020	.016	.760
1971	.003	.000	.000	.020	.016	.750
1970	.000	.000	.000	.020	.016	.750
1969- and later	.000	.000	.000	.000	.000	.750

Table 10-15

1995 Urban Diesel Particulate Emissions Under the Base Scenario (metric tons)*

	Best Estimate Diesel Sales	No Growth Diesel Sales	% Reduction from Best Estimate
LDDV	19,700	10,100	49
LDDT	11,700	3,400	71
MDDV/LHDDV	4,800	2,700	44
HHDDV	36,100	30,300	13
TOTAL	72,200	46,500	36

---

* Assumes an LDDV NOx standard of 1.0 g/mi and an LDDT NOx standard of 1.2 g/mi.

Reference

1. "An Evaluation of Particulate Levels Occurring Under 1.0/1.2 g/mi NOx Standards for LDDVs and LDDTs," R. Kanner, Technical Report, SDSB, U.S. EPA.

2. "Historical and Projected Emissions, Conversion Factor, and Fuel Economy for Heavy-Duty Trucks: 1962-2002," prepared by Energy and Environmental Analysis, Inc., for the Motor Vehicle Manufacturers Association, December 1983.

APPENDIX

Table A-1

Estimates of the Potency of Organic Extracts of Diesel Exhaust and Related Environmental Emissions - Harris*

Emissions Extract	Tumor Initiation in SENCAR Mice (papillomas/mouse per mg extract at 27 weeks)	Enhancement of SA7 Viral Transformation in Syrian Hamster Embryo Cells (tranformations/2x10^6 cells per ug extract/ml)
Coke Oven	2.101 (0.090)	0.859 (0.089)
Roofing Tar	0.535 (0.024)	2.066 (0.363)
Caterpillar 3304 Diesel Engine	0.011 (0.009)	0.039 (0.023)
Nissan Datsun 220-C Diesel Engine	0.528 (0.023)	0.645 (0.095)
Oldsmobile 350 Diesel Engine	0.156 (0.034)	0.067 (0.023)
Volkswagen Turbocharged Rabbit Diesel Engine	- -	0.128 (0.023)
Benzo(a)pyrene Positive Control	85.28 (2.71)	540. (21.9)

---

* Table excerpted from Reference 1. Maximum likelihood estimates of slope of linear dose response model based upon Poisson distribution of positive responses. Asymptotic standard errors in parentheses.

Table A-2

Estimates of Potency of Organic Extracts of
Diesel Exhaust and Related Environmental Emissions
in L5178Y Mouse Lymphoma Mutagenesis Assay - Harris*

	Average Mutant Colonies/10^6 Survivors Per ug Extract/ml	
Emissions Extract	- Metabolic Activation	+ Metabolic Activation
Coke Oven	0.726 (0.152)	9.963 (0.734)
Roofing Tar	0.311 (0.121)	9.556 (1.547)
Caterpillar 3304 Diesel Engine	0.156 (0.038)	0.049 (0.021)
Nissan Datsun 220-C Diesel Engine	1.662 (0.509)	1.869 (0.485)
Oldsmobile 350 Diesel Engine	0.270 (0.117)	0.764 (0.109)
Volkswagen Turbocharged Rabbit Diesel Engine	2.545 (0.402)	1.012 (1.200)

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* Table excerpted from Reference 1. Maximum likelihood estimates of slope of linear dose response model based upon Poisson distribution of positive responses. Asymptotic standard errors in parentheses.

Table A-3

Summary of Results From Mutagenesis and Carcinogenesis Assays Using Diesel Exhaust Particle Extracts and Other Surrogate Substances - Lovelace*

Surrogate Exposure	AMES ASSAY		HGPRT GENE LOCUS ASSAY	MOUSE LYMPHOMA CELL ASSAY		BALB/3T3 CELL ASSAY				HAMSTER EMBRYO CELL ASSAY	SENCAR MOUSE SKIN ASSAY	
	Revertants Per 100 ug Extract		Mutants/10^6 Survivors Per ug/ml Extract	Mutants/10^6 Survivors Per ug/ml Extract		Mutation Frequency X 10^5		Transformation Frequency X 10^5		Transformations Per 2 X 10^6 Cells Per ug/ml Extract	Papillomas per Mouse at 1 mg	Carcinoma per Mouse at 1 mg
	With S-9	W/O S-9	With S-9	With S-9	W/O S-9	With S-9	W/O S-9	With S-9	W/O S-9			
Diesel Vehicle Exhaust	300	500	0.2	1.9	1.7	0.5	0.51	-0.5	0.3	0.6	0.25	0.05
Coke Oven Emissions	250	10	6	10	0.7	n.a.	8.3	2.5	2.1	0.86	2	0.1
Roofing Tar Vapors	100	0	6	10	0.3	1.7	3.1	1.1	0.6	2.1	0.4	0.1
Cigarette Smoke Condensate	100	0	0.06	0.5	0.6	n.a.	n.a.	n.a.	n.a.	0.6	0.2	0.1
Urban Soot	100	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.

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* This table was excerpted from Reference 15.

Table A-4

Summary of Results for Human Lung Cancer Annual Unit Risks - EPA* (risk/ug organics/m^3)

Emission Source	Lower Limit	Best Estimate	Upper Limit
Coke Oven[a]	6.6×10^{-6}	1.2×10^{-5}	2.0×10^{-5}
Roofing Tar[a]	1.3×10^{-6}	4.7×10^{-6}	9.4×10^{-6}
Cigarette Smoke[b]	1.7 X 10-8	2.9×10^{-8}[c]	4.9×10^{-8}

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* This table was excerpted from Reference 16 in which lifetime risks were presented. These risks have been converted to annual risks by dividing by the median lifespan (76.2 years).

[a] 95 percent confidence intervals for linear model.

[b] Bounds from linear and quadratic model.

[c] Geometric mean of the limits.

Table A-5

Summary of Dose-Response Slopes for Emission Extracts Short Term In-Vitro Bioassays - EPA*

	Mutation in L5178Y Mouse Lymphoma Cells[b]		SCE in CHO Cells[c]		Ames Salmonella Typhimuriium TA98[d]	
Sample	(-MA)[a]	(+MA)	(-MA)	(+MA)	(-MA)	(+MA)
Human Carcinogens:						
Coke Oven	0.71	12.	0.41	0.03	0.7	1.1
Roofing Tar	0.39	17.	0.12	0.02	Neg[e]	0.86
CSC	0.39	0.79	0.12	0.08	Neg	0.57
Diesels:						
Nissan	4.2	2.9	0.30	0.071	11.	13.
Volkswagen Rabbit	0.98	0.72	0.075	0.030	3.8	3.0
Oldsmobile	1.2	1.3	Neg	0.017	2.2	1.5
Caterpillar	0.25	0.063	0.011	Neg	0.38	0.31

---

* This table was excerpted from Reference 16.

[a] MA = Metabolic Activation.

[b] Mutation frequency (TK mutants/10^6 surviving cells)/ug/ml.

[c] SCE/cell/ug/ml, (-MA) was a 21.5-h exposure and (+MA) was a 2-h exposure.

[d] Revertants/ug (from simple linear regression analysis).

[e] Neg = Negative, i.e., no response.

Table A-6

SENCAR Mouse Skin Tumor Initiation and Complete Carcinogenesis by Emission Extracts*

Sample	Skin Tumor Initiation		Skin Cancer
	Multiplicity Data (papillomas/ mouse at 1 mg)	Incidence Data (dose in mg yielding 25% mice with papillomas)	Incidence Data (dose in mg yielding 25% mice with carcinomas)
Human Carcinogens:			
Coke Oven Topside	2.1[d] (1.0)	0.16[b] (1.0)	--
Roofing Tar	0.41[d] (0.20)	0.71[b] (0.22)	4.0[c]
CSC	0.0024[a,e] (0.0011)	92. (0.0017)	--
Diesels:			
Nissan	0.59[d] (0.28)[f]	0.61[b] (0.26)	[g]
Volkswagen Rabbit	0.24[a]	--	--
Oldsmobile	0.31[a]	--	
Caterpillar	Neg[h]	--	--
Gasoline Catalyst			

---

* This table was excerpted from Reference 16.

[a] Values based directly on papilloma multiplicity data at 1 mg.

[b] Values based on statistical analyses of the papilloma incidence data by log-Probit model with background correction.

[c] Values based on carcinoma incidence data.

[d] Values based on statistical analyses of the papilloma multiplicity data by a Poisson model with background correction.

[e] Nesnow, Triplett, and Slaga, unpublished observations.

[f] Values in parentheses are normalized to the coke oven topside sample.

[g] Nissan produced carcinomas in 4 percent of the mice at the 4-mg/week dose level.

[h] Neg = negative levels.

Table A-7

Comparison of Relative Potencies of Emission Extracts in Several Bioassay Systems - EPA*

Emission Source	Mouse Skin Tumor Initiation	Mouse Skin Cancer	Mutation in Mouse Lymphoma Cells(+MA)[a]	Mutation in Ames TA98 (+MA)[a]	Human Lung Cancer
Coke Oven[b]	1.00	1.00	1.00	1.00	1.00
Roofing Tar	0.20	0.20	1.40	0.78	0.39
Cigarette Smoke	0.0011	NT[c]	0.066	0.52	0.0024
Nissan Diesel	0.29	0.10	0.24	12.00	ND[d]

---

* This table was excerpted from Reference 16.

[a] (+MA) = With Metabolic Activation.

[b] Absolute value of annual unit risk is 1.2×10^{-5}.

[c] NT = Not Tested.

[d] ND = No Data.

Table A-8

Comparison of Relative Potencies of Emission Extracts in Several Bioassay Systems - EPA*

Diesel Source	Mutation in Mouse Lymphoma Cells (+MA)[a]	SCE in CHO Cells (+MA)[a]	Mutation in Ames TA98 (+MA)[a]	Net Relative Potency
Nissan[b]	1.00	1.00	1.00	1.00
Volkswagen	0.25	0.42	0.23	0.30
Oldsmobile	0.45	0.24	0.11	0.27
Caterpillar	0.022	NEG[c]	0.023	0.015

---

* This table was excerpted from Reference 16, except for the last column, which was derived from the final risk estimates of Table 3.

[a] (+MA) = with metabolic activation.

[b] Absolute value of annual unit risk is 0.58×10^{-5}.

[c] NEG = negative (i.e., no response).

Table A-9

Ames Test Results on Diesel Vehicles (using TA-98 strain)

Studv/Reference	Vehicles	Mean Specific Activity (revertants/ug SOF)	
		(-MA)	(+MA)
Landman/Wagner[17]	In-use Oldsmobiles (40k miles)	13.7±7.4	8.22±2.75
	Peugeot	6.46±4.49	2.08±0.46
	Prototypes (Mercedes-Benz, Peugeot)	11.1±3.9	8.66±4.59
McMahon, et.al.[18]	VW Rabbit	5.84±6.6	5.26±1.57
Hyde, et.al.[19]*	GMC	1.24±0.52	--
	VW Rabbit	1.10±0.77	--
	Mercedes-Benz	0.97±0.27	--
	Others	0.97±0.24	--
Claxton[20]	Diesel Engine	4.35±0.64	--
	Diesel Vehicles	6.96±4.06	--
	Overall Mean =	5.27	6.06

---

* All vehicles in this study were in-use diesels.

Table A-10

Total Particulate and SOF Emission Rates for LDDVs With and Without Ceramic Traps

Manufacturer		Without Trap		With Trap		+ % Change	
Vehicle	Trap	TPM	SOF	TPM	SOF	TPM	SOF
[22] Ford	-- (1)	436	309	192	173	-56	-44
		303	138	63.6	56.6	-79	-59
		436	309	91.6	83.4	-79	-73
		303	138	75.8	63.5	-75	-54
[23] Mercedes	Corning (2)	0.25	0.018	0.050	0.004	-80	-78
		--	--	--	--	-90	-75
[23] Datsun	NGK (2)	0.17	0.045	0.040	0.012	-76	-73
[24] GM	Corning	--	--	--	--	-89	-77
						-97	-92
					Average =	-80	-69

[ ] Reference
(1) mg/mi
(2) g/km

Table A-11

Total Particulate and SOF Emission Rates for HDDV and HDDEs With and Without Ceramic Traps

Manufacturer		Without Trap		With Trap		± % Change	
Vehicle	Trap	TPM	SOF	TPM	SOF	TPM	SOF
[25]Caterpillar							
Engine	Corning	1.047	0.932	1.039	0.972	-0.76	+4.3(a)
	(1)	0.482	0.377	0.173	0.137	-64.1	-63.7
		0.241	0.038	0.015	0.002	-93.8	-94.7
		0.795	0.026	0.022	0.0005	-97.2	-98.1
		0.792	0.133	0.022	0.006	-97.2	-95.5
		1.864	0.969	0.139	0.067	-92.5	-93.1
[26] DDAD							
Coach Engine	Corning	0.70	0.20	0.15	0.11	-78.6	-45.0
	(1)	0.86	0.49	0.28	0.24	-67.5	-51.0
		0.70	0.39	0.30	0.25	-57.1	-35.9
		0.75	0.40	0.29	0.25	-61.3	-37.5
		0.78	0.54	0.25	0.20	-68.0	-63.0
[26] GMC	Corning	5.48	0.46	0.589	0.073	-89.3	-84.1
Coach	(2)	4.24	0.28	0.313	0.035	-92.5	-87.5
Vehicle		4.4	0.31	0.350	0.040	-92.1	-87.1
		6.2	0.41	0.430	0.049	-93.1	-88.1
				Average =		-76.3	-68.0

[ ] Reference
(1) g/kw-hr
(2) g/km
(a) EPA Test Mode 3

Table A-12

SOF as Percent of Total Particulate for LDDVs With and Without Ceramic Traps

Manufacturer		Without	With	±
Vehicle	Trap	Trap	Trap	Percent Change
[22] Ford	--	70.9	90.1	+19.2
		45.5	89.0	+43.5
		70.9	91.0	+20.1
		45.5	83.8	+38.3
[23] Mercedes	Corning	7.2	8.0	+0.8
[23] Datsun	NGK	26.5	30.0	+3.5
[24] GM	Corning	25.5	55.1	+29.6
		25.0	57.0	+32.0
			Average =	+23.4

[ ] Reference

Table A-13

SOF as Percent of Total Particulate for HDDV and HDDEs With and Without Ceramic Traps

Manufacturer				
Vehicle	Trap	Without Trap	With Trap	± Percent Change
[25] Caterpillar Engine(a)	Corning	89.0	93.6	+4.6
		78.2	79.2	+1.0
		15.8	13.3	-2.5
		3.3	2.3	-1.0
		16.8	27.3	+10.5
		52.0	48.2	-3.8
[26] DDAD Coach Engine	Corning	28.9	75.0	+46.1
		56.8	84.4	+27.6
		56.1	82.7	+26.6
		56.2	82.9	+26.7
		64.6	81.8	+17.2
[26] GMC Coach Vehicle	Corning	8.4	12.4	+4.0
		6.6	11.2	+4.6
		7.1	11.4	+4.3
		6.6	11.4	+4.8

Average = +11.4

(a) Caterpillar test modes: EPA 3, 4, 5, 9, 10, 11.

Table A-14

Gaseous HC Emissions for LDDVs With and Without Ceramic Traps

Manufacturer		Gaseous HC Emissions		
Vehicle	Trap	Without Trap	With Trap	± Percent Change
[22] Ford (1)	--	0.99	0.90	-9.1
		0.61	0.60	-1.6
		0.99	0.63	-36.4
		0.61	0.47	-22.9
[17] Toyota (1)	NGK	0.405	0.313	-22.7
		0.223	0.161	-27.8
[17] Mercedes (1)	Corning	0.266	0.230	-13.5
		0.092	0.091	-1.1
[23] Mercedes (2)	Corning	0.0970	0.0645	-33.5
[23] Datsun (2)	NGK	0.20	0.18	-10.0
[24] GMC (3)	Corning	38	38	0
		26	26	0
		26	25	-3.9
		27	26	-3.7
		28	28	0

Average = -12.4%

[ ] Indicates reference
(1) g/mi
(2) g/km
(3) ppm-C3

Table A-15

Gaseous HC Emissions for HDDV and HDDEs With and Without Ceramic Traps

Manufacturer		Gaseous HC Emissions		
Vehicle	Trap	Without Trap	With Trap	± Percent Change
[25] Caterpillar Engine	Corning(1)	1.917	2.152	+12.3 (a)
		0.771	0.591	-23.4
		0.253	0.178	-29.6
		0.265	0.111	-58.1
		1.305	0.510	-60.9
		2.926	2.077	-29.0
[26] DDAD Coach Engine	Corning(1)	1.64	1.68	+2.44 (b)
		1.85	1.88	+1.62
		1.91	1.89	-1.05
		1.90	1.89	-0.53
		1.93	1.83	-5.18
[26] GMC Coach Vehicle	Corning(2)	1.78	1.10	-38.2
		1.52	1.01	-33.6
		1.56	1.02	-34.6
		2.25	1.47	-34.7
			Average =	-22.17%

[ ] Reference
(1) g/kh-hr
(2) g/km
(a) EPA Mode 3 (test condition)
(b) 13-Mode composite

Table A-16

Gaseous HC Emissions During Durability Test of Corning Trap on Mercedes 300SD [reference 23]

	HC Emissions (g/km)		
VMT (1000s km)	Without Trap	With Trap	± Percent Change
0	0.11	0.080	-27
8	0.11	0.070	-36
16	0.12	0.080	-33
24	0.09	0.060	-33
32	0.08	0.070	-12
40	0.09	0.055	-39
48	0.09	0.070	-22
56	0.10	0.053	-47
64	0.09	0.060	-33
72	0.09	0.041	-54
80	0.10	0.068	-32
			Average = -33.5%

Table A-17

Ames Test Bio-Activity Data on SOF for LDDV With and Without Ceramic Trap [reference 27]

Test Condition	Without Trap		With Trap		± % Change	
Trap; Engine	(a)per mass	per(b) VMT	per mass	per VMT	per mass	per VMT
Clean Trap, A	1.9	5.9	3.7	6.4	+94.7	+8.5
Clean Trap, B	2.0	2.8	5.1	3.0	+155	+7.1
Loaded Trap, A	1.9	5.9	3.6	3.0	+89.5	-49.2
Loaded Trap, B	2.0	2.8	4.3	2.9	115	+3.6
				Average =	+113.6	-7.5

(a) Bio-activity per mass, units of revertants/ug SOF.
(b) Bio-activity per VMT, units of revertants/mile x 10^{-5}.

Table A-18

Ames Test Bio-Activity Data on SOF for HDDE With and Without Ceramic Trap [reference 25]

EPA Test Mode	Without Trap		With Trap		± % Change	
	(a) per mass	per (b) work	per mass	per work	per mass	per work
3	0.408	363	0.392	365	-3.92	+0.55
4	0.251	92	1.404	193	+459	+110
5	1.319	54	1.524	3.3	+15.5	-94.0
9	0.851	22	2.482	2.6	+192	-88.2
10	0.762	101	1.620	10.7	+113	-89.4
11	0.357	325	1.250	87	+250	-73.4
				Average =	+170.9	-39.1

(a) Bio-activity per mass, units of revertants/ug SOF

(b) Bio-activity per work, units of revertants/kw-hr

Table A-19

Temperature Range Distribution of SOF for HDDE With and Without Ceramic Trap [reference 26]

Percent Mass Released	Boiling-Point Temperature,°C		± % Change
	Without Trap	With Trap	
IBP	307	340	+10.8
10	391	396	+1.28
20	412	418	+1.46
30	432	435	+0.69
40	452	450	-0.44
50	474	465	-1.90
60	503	480	-4.57
70	542	499	-7.93
80	607	530	-12.7

Table A-20

BaP Emission Rates for HDD Engine and Vehicle With and Without Corning Ceramic Trap [reference 26]

Test Conditions		BaP Emission Rate			± % Change
		Without Trap	With Trap		
Coach (1) Engine:	7-mode	0.04	0.12		+200
	Transient	0.08	0.28		+250
	Bus Cycle	0.11	0.11		0.0
				Average =	150%
In-Service Coach (2):	Transient	0.050	0.055		+9.6
	Bus Cycle	0.008	0.022		+175
				Average =	92%

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(1) BaP, ug/kw-hr

(2) BaP, ug/km

Table A-21

Trap Removal-Efficiencies of Carcinogens With Respect to Total Particulate

		Average % Change		Ratio of Percent Changes	
Carcinogen	Study	Carcinogen	TPM	Average	Range
SOF	Ford	-56.5	-77.0	.74	.72-.75
	Mercedes	-76.5	-85.0	.91	.83-.98
	Datsun	-73.3	-76.5	.96	.96
	GM	-84.5	-93.0	.91	.87-.95
	Caterpillar	-89.0	-89.0	1.0	.98-1.01
	DDAD Engine	-46.5	-66.5	.70	.57-.93
	GMC Coach	-86.7	-91.8	.94	.94-.95
Mutagens	Ford	-7.5	-72.3	.10	.10
	Caterpillar	-86.3	-89.0	.91	.79-1.0
BaP	DDAD Engine	+150	-66.5	-2.53	(-4.1)-0
	GMC Coach	+92	-91.8	-.99	(-.104-(-1.88)